Ecological evaluation of restored former sewage channels in the urbanised Emscher catchment

Inaugural-Dissertation
zur
Erlangung des Doktorgrades
Dr. rer. nat.

der Fakultät für
Biologie
an der

Universität Duisburg-Essen

vorgelegt von
Caroline Winking
geboren in Coesfeld

April 2015

Die der vorliegenden Arbeit zugrunde liegenden Experimente wurden in der Fakultät für Biologie in der Abteilung Aquatische Ökologie der Universität Duisburg-Essen durchgeführt.

1. Gutachter: Prof. Dr. Daniel Hering

2. Gutachter: PD Dr. Jost Borcherding

Vorsitzender des Prüfungsausschusses: Prof. Dr. Peter Bayer

Tag der mündlichen Prüfung: 21.08.2015

Abbreviations

DIN	German Industry Standard (Deutsche Industrienorm)
Dispersal class A	Hololimnic, wingless taxa without the ability for aerial dispersal
Dispersal class B	Winged adult stage, low dispersal capabilities, habitat specialists
Dispersal class C	Winged adult stage, high dispersal capabilities, habitat specialists
Dispersal class D	Winged adult stage, low dispersal capabilities, habitat generalists
Dispersal class E	Winged adult stage, high dispersal capabilities, habitat generalists
Dispersal class x	Taxa without classification to a dispersal class
EG	European Commission (Europäische Gemeinschaft)
EP	Ecological Potential
GEP	Good Ecological Potential
GIS	Geo Information System
HMWB	Heavily Modified Water Bodies
MHS	Multi-Habitat-Sampling
NMS	Non-metric Multidimensional Scaling(s)
NRW	North-Rhine Westphalia
PCA	Principal Component(s) Analysis
PHQ	Physical Habitat Quality
RC	Restored site, connected to near-natural upstream section or tributary
RU	Restored site, <u>not</u> connected to near-natural upstream section or tributary
RQ	Recolonisation Quotient
SB	Source sites within Boye catchment
SO	Source site outside Boye catchment
WFD	Water Framework Directive

Contents

List of Figures

List of Tables

1 Background and scope of the thesis

Urban streams and its response to restoration are in the focus of this thesis. The typical urban stream is influenced by various pressures like pollutants, missing riparian buffers, storm waters, contaminations, a modified hydro-morphology and a modified hydro-regime (Beavan et al. 2001). Often, long distances are piped, the lateral development is limited due to buildings or roads and the remaining stream sections are isolated (Bernardt & Palmer 2007). As a logic consequence, the biological assemblages of these streams are degraded (Paul & Meyer 2001; Blakely et al. 2006). Restoration projects to improve the biological status in urban streams are often complex and expensive. Nevertheless, the number of morphological restorations and water quality improvements in urban streams has strongly increased in the last two decades (Walsh et al. 2005). Therefore, an investigation of what happens to urban streams after water quality and morphology have been restored is necessary.

A good study area to investigate restored urban streams is the highly urbanised Emscher catchment in western Germany. Even compared to other urban areas most streams of the Emscher catchment are degraded in an unusual way and have a unique history: they have been used as open sewers for partly about one century, reducing the former fauna to only Oligochaeta which endured the constant wastewater.
The Emscher catchment is located in the “Ruhr Metropolitan Area”, a region with a population density of about 2700 inhabitants per km² and a total of 5 million inhabitants of whom 2.4 million inhabitants are living in the Emscher catchment. The Emscher catchment itself has a size of 865 km² of which 48% is covered by intensive used build-up settlement areas (EGLV 2015). About 200 years ago, the stream catchment was characterised by meandering, braiding gravel or sand streams with a low slope and regular water flow. Since the 1860s and the beginning of the industrialisation, the streams were increasingly used to discharge the untreated wastewater of industries, agriculture and coal mining. For this purpose, the Emscher and parts of its tributaries were straightened. Further, the natural run-off capacity and the hydro-morphological conditions were disturbed by mining related subsidence. Occasional high flood situations caused extremely bad hygienic situations for the inhabitants of the region. To solve this problem, a regional water broad - the

Emschergenossenschaft (EGLV) - was founded in 1899. Under their supervision an open channel system was constructed to discharge the wastewater of the region. An underground sewer system was not possible due to mining-related subsidence. In total, 350 stream kilometres out of the existing 640 stream kilometres in the catchment were thereto laid in concrete channels. They partly remained in this situation for about 100 years (Fig. 1.1a). Around 290 km were left un-channelised and in a more or less near-natural morphological state (Fig. 1.1b).

In the early 1990s, with the end of coal mining and expected subsidence, the EGLV decided to restore the 350 km of channelised streams (Fig. 1.1c). Currently 123 km of 350 km have already been restored (Fig 1.1d,e and Fig. 1.2) and 85 km of 350 km are already wastewater free (Fig 1.1f). Restoration measures mainly conducted the removal of the concrete slaps, the separation of the wastewater into underground sewers and the remodelling of the stream bed and the streams surroundings. Due to the deep incision of the channels and the main river Emscher following the straightening and the groundwater depletion it is often not possible to restore a primary floodplain, but solely a secondary floodplain. Additionally, the self-dynamic development is limited at many sites. Whenever possible, riparian buffers were created and vegetation succession was allowed (EGLV 2015).

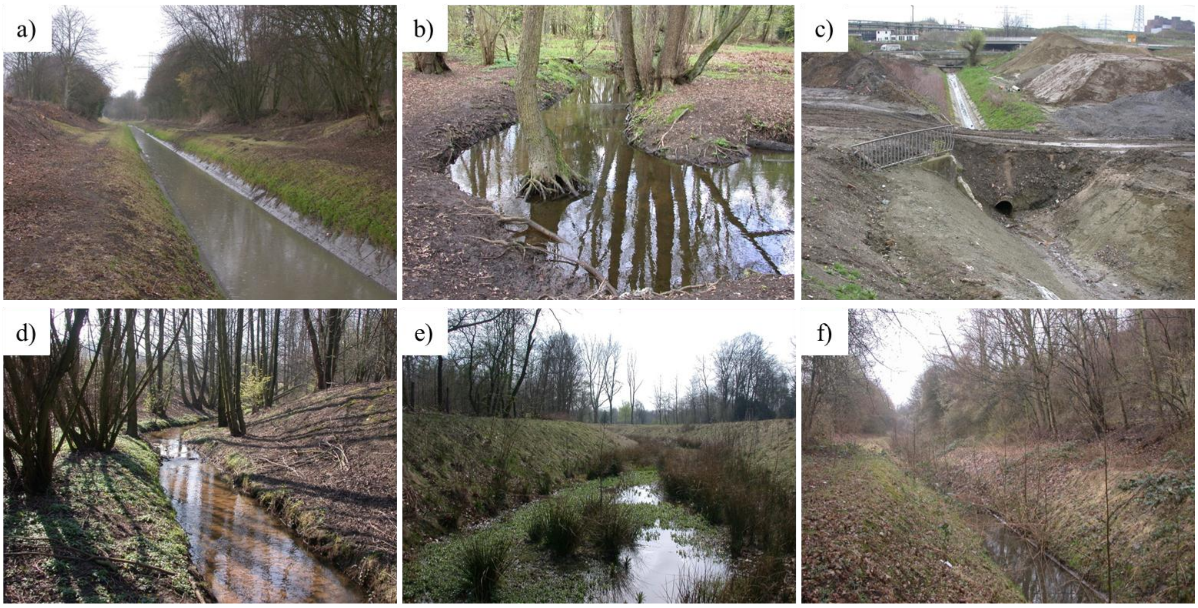

Figure 1.1 Example pictures of the status prior to restoration and the different states of restoration. a) wastewater channel prior to restoration (Kirchschemmsbach); b) never channelised, near-natural section (Spechtsbach); c) section during a restoration phase (Goldammer Bach); d) restored site 20 years after restoration (Laeppkes Muehlbach); e) restored site one year after restoration (Haarbach); f) wasterwater free concrete channel, removal of the concrete slaps is planned (Wittringer Bach).

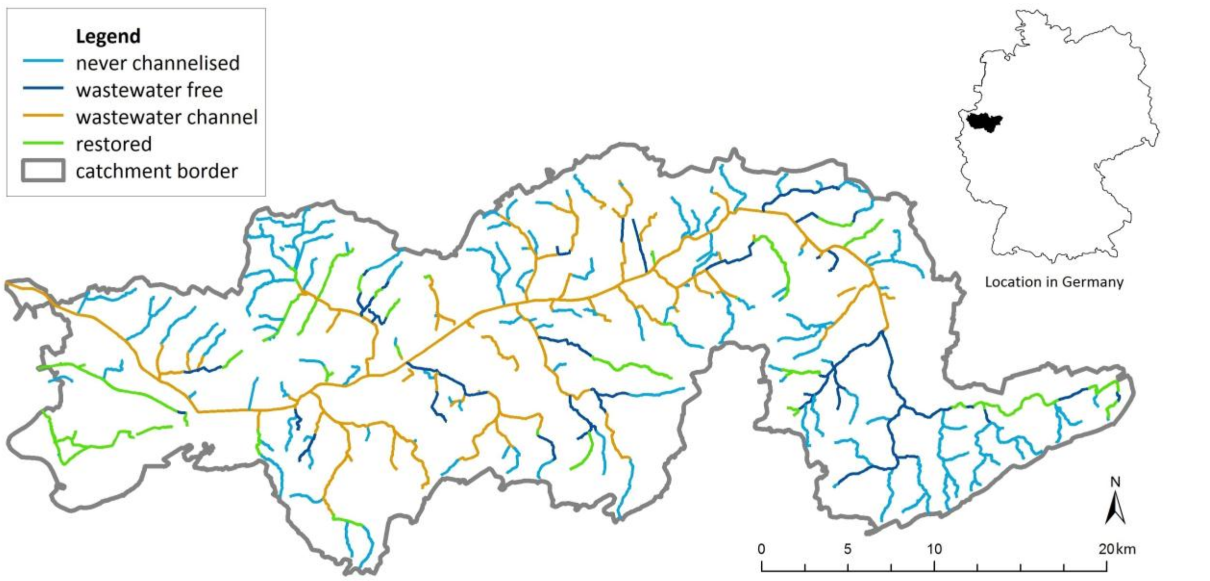

Figure 1.2 The Emscher catchment with the status of restoration (status of December 2014) and its location in Germany.

Since the European Water Framework Directive (WFD) (European Commission 2000) entered into force, it has had major influences on the applied river management. The WFD aims at improving the chemical and biological quality of European waters (including coastal waters, surface waters and the groundwater) with the goal to reach a “Good Ecological Status” for both until 2015 (or under certain circumstances until 2021 or 2027). The specifications of the WFD were also integrated in the restoration of the Emscher system.

For the assessment of streams according to the WFD, amongst others, benthic invertebrates are used. Benthic invertebrates are small creatures that live on the stream bottom or buried in the substrate, e.g. insect larvae, mussels, snails and crustaceans. The species pool on benthic invertebrates, which is used for stream assessment, includes more than 1,000 species and higher taxa (Haase et al. 2011), which are adapted to very different habitats, food sources and oxygen levels, which in turn influence their distribution (Malmquist 2002). In summary, these species give a comprehensive picture of the overall status of a stream (Hering et al. 2004). Therefore, this organism group (and its development) is in the focus of this thesis.

In Germany stream assessment is based on a standardised sampling: the collected taxa are identified and the resulting taxa list is used to calculate various indices, whose values are then compared with expected values of streams in natural or near-natural condition. Finally, the assessment is drawn by this comparison. In Germany 24 reference stream types reflect the topographical, geological and sedimentological differences in stream characteristics (Lorenz et al. 2004; LAWA 2004) and are used as reference conditions. Depending on the stream type different indices and class boundaries are used. In the case of “heavily modified water bodies” (HMWB) where human use is in the foreground, the waters are assigned to HMWB-types, depending on the predominant use of which has led to the designation of heavily modified water bodies (e.g. flood protection, land drainage or urban use). Their assessment is orientated on the community, which would be present after the maximum possible improvement (Good Ecological Potential); in practice, it is carried out using similar methods as for natural waterbodies, but with less stringent limits (LAWA 2013).

As soon as a stream section is wastewater free and morphologically restored recolonisation by benthic invertebrates starts. They migrate from surrounding streams which act as recolonisation sources. Anthropogenic barriers like houses and factories can impede recolonisation. In assumption, various factors acting at different scales influence the recolonisation by benthic invertebrates and hence, the restoration success of streams. Thereunder are predominantly the following factors: recolonisation potential, meaning recolonisation sources in the surrounding (e.g. Sundermann et al. 2011a; Tonkin et al. 2014), meta-population dynamics (Heatherly et al. 2007), the species dispersal capability (Cañedo-Argüelles et al. 2015), the restored streams environmental conditions and landscape context e.g. micro-habitats, land use, riparian vegetation (Hughes et al. 2008; Huang & Guo 2014; Reynolds et al. 2013) and succession processes (McCook 1994). The field of restoration research as a science is still young (Palmer et al. 2014) and a lot of research studies already aimed at the response of benthic invertebrates to stream restoration in general (e.g. Jähning et al. 2010; Palmer et al. 2010; Tullos et al. 2009; Violin 2011). In urban streams however the knowledge about invetebrates’ response to restorations is still limited (Kenney et al. 2012). Furthermore, in terms of river basin management restoration projects lack evaluations whether they can successfully enhance habitat structure or support the stream assemblage of reference sites (Violin et al. 2011).
This thesis tries to expand the scientific basis on urban stream restoration and their restoration success. Indicators of the WFD were used and new indicators developed. To lay the foundation and as a general overview on the ecological assessment, the Ecological Potential according to the WFD was calculated for restored streams in the whole Emscher catchment. Thereby, environmental parameters limiting or enhancing the Ecological Potential were evaluated and recommendations for future restoration projects in urban areas were derived (chapter 2). As recolonisation processes are of fundamental importance for the restoration success, analyses of the dispersal of taxa colonising the restored streams, and further

recolonisation patterns were applied in restored streams of an Emscher sub-catchment - the catchment of the Boye (chapter 3). Moreover, the catchment of the Boye was analysed for two consecutive years. In the second year the research focus switched from recolonisation patterns to the temporal process of recolonisation and to environmental parameters steering succession processes (chapter 4).

In summary, the general aim of the ecological evaluation of the restored streams of the Emscher catchment is to analyse the success of the different stream restorations on several point of views, in order to optimise future restoration projects in urban streams. Finally, this thesis gives implications for the design of urban stream restoration projects and consecutive monitoring programmes.

The three mentioned chapters have their own detailed introduction and sub-chapters with methods, results and discussion. Further, a summary and an overall conclusion are given at the end of the thesis.

2 Restoration of a river system in an urban area: towards the Good Ecological Potential of former sewage channels

2.1 Introduction

Urban streams differ in many ways from streams and rivers in the open landscape: their lateral development is limited due to roads and buildings, their hydro-morphology and water quality are affected by a variety of stressors and often long sections are piped so that the remaining sections are isolated (Bernardt & Palmer 2007). All this have repercussions on the benthic assemblages that are mostly degraded (Suren 2000; Paul & Meyer 2001; Blakely et al. 2006) and characterised by missing stress-sensitive species (Coles et al. 2012).

In Europe, according to the Water Framework Directive (WFD) (European Commission 2000) urban streams are mostly classified as "heavily modified water bodies" (HMWB). These water bodies are subject to stream-specific anthropogenic pressures, which cannot be removed due to high social or economic costs. Therefore, the "Good Ecological Status", the ambitious objective of the WFD, is usually not attainable and replaced by the "Good Ecological Potential" as a management goal. The German federal water association (Länderarbeitsgemeinschaft Wasser, LAWA) has recently published an extensive methodological guide to assess the Ecological Potential of heavily modified water bodies (LAWA, 2013).

The Emscher catchment is a prototype for urban streams and rivers. The catchment is located in the Ruhr Metropolitan Area (western Germany) with more than 5 million inhabitants, 2.4 million of whom are living in the Emscher catchment. Even compared to other urban regions most streams of the Emscher catchment are unusually degraded: laid in concrete channels, they served as open sewers for about 100 years and transported the wastewater of the region. Due to the mining-induced subsidence of the surface it was, for a long time, not possible to build underground sewers. The Emscher and the downstream sections of its tributaries have therefore been used as above-ground sewers of a total length of about 350 km. Furthermore, the soils of the Ruhr Metropolitan area are frequently contaminated, which can affect the streams with pollutants of all kinds. Road wastewater, storm water overflows and sewage overflows also contribute to a complex stress situation.

Since the coal mining came to an end and subsidence is not expected anymore, the regional water board - the Emschergenossenschaft - has started to restore the Emscher and its tributaries with the aim to reach the Good Ecological Potential. Underground sewers were built to transport the wastewater, the concrete shells were removed, and the stream beds and the riparian areas were restored. The total investment amounts to 4.5 billion €, which is mainly used for the underground sewers and purification plants, and to a lesser degree for the morphological restoration of the river system. Until now, about 123 km (status of December 2014) of the total of 350 km have already been restored (EGLV 2015).

Contrary to other stream restoration projects, no higher organisms occurred in the restored former sewage channels; for decades only Oligochaeta endured the wastewater (Winking et al. 2014). Thus, an entirely novel benthic invertebrate community developed in the new streams, which may, however still be inhibited by water pollution and anthropogenic barriers. Similarly to other restored rivers sections, pressures acting at larger spatial scales might shape the benthic assemblages more strongly as compared to pressures acting at the site scale (Kail & Wolter 2013; Black et al. 2004; Hughes et al. 2008). In other studies, however, site scale pressures are described of equal influence for the assemblages than those acting at the catchment scale (e.g. Feld & Hering 2007, Verdonschot 2009). Additionally, Kail & Hering (2009) and Lorenz & Feld (2013) described the hydro-morphological status of the adjacent upstream river catchment as an important factor influencing the benthic assemblages and the ecological status of restored streams. In summary, the environmental parameters that affect benthic invertebrate assemblages are not fully understood and it is unclear which are most relevant for the development of restored urban streams.

In the present study, we investigated the Ecological Potential of restored streams in the Emscher catchment. We particularly addressed the three following questions:

(Q1) What is the ecological status of the restored streams in the Emscher catchment?

(Q2) Which environmental parameters influence the Ecological Potential of the restored streams?

(Q3) Which recommendations arise for future restoration projects of the Emscher catchment and of urban streams in general?

2.2 Methods

2.2.1 Study streams

The Emscher is a right tributary of the River Rhine, flowing from Holzwickede in a westerly direction through the densely populated Ruhr Metropolitan area. The catchment area comprises 865 km²; the Emscher has a length of 82 km. Although the catchment area is mainly urbanised (48 %, EGLV 2015), many upstream stream sections flow through rural or forested areas. Here, the streams were never used as sewers and remained in a near-natural stage. The stream sections in the Emscher catchment can be divided in the following categories (Figure 2.1):

- Open sewer (unrestored) sections, which still transport wastewater (142 km).
- Wastewater free sections, which transports the water in concrete channels (85 km).
- Restored sections (123 km): they formerly transported wastewater in aboveground concrete channels; now the wastewater flows in underground sewers. The concrete shells were removed, and - if so authorised by the topography - a natural stream bed and riparian areas were modeled.
- Near-natural sections which were never used as open sewers (about 290 km). These sections usually are upstream sections of the catchment; in contrast to the restored sections they were always colonised by benthic invertebrates and may now act as recolonisation sources.

2.2.2 Data base and data processing

<u>*Benthic invertebrates*</u>

The data basis of our analyses were benthic invertebrates´ taxa lists of restored sites derived from the monitoring of the regional water board (Emschergenossenschaft, EGLV), the State Agency for Nature, Environment and Consumer Protection NRW (LANUV NRW) and from own samplings. We considered only samples taken in the spring season. Samples from the years 1994 to 2013 were considered, which resulted in 248 taxa lists from 48 sampling sites in 13 streams (Table 2.1).

The samples were either taken according to the German standard multi-habitat-sampling protocol (Perlodes-method, Meier et al. 2006) or according to the less specifically described method according to DIN 38410 (1987). To counteract a possible difference in identification precision, the taxa lists were harmonised before analysis (Nijboer & Schmidt-Kloiber 2004).

The Ecological Potential (EP) of the 248 taxa lists was calculated with the software PERLODES (ASTERICS software, version 4.03), whereas the HMWB - type "Flood Protection and Urbanisation (with foreland)" was used.

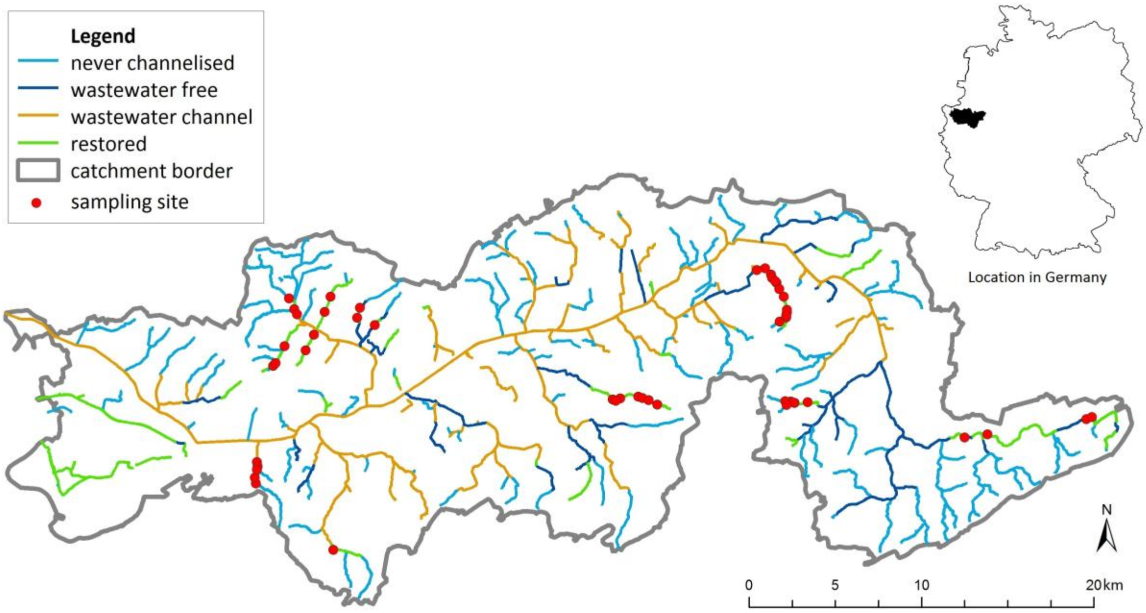

Figure 2.1 The sampling sites and their location in the Emscher catchment.

Table 2.1 Overview of the investigated streams with the number (#) of sampling sites per stream, the number (#) of invertebrates´ taxa lists per stream, and the sampling period.

Water body name	# Sampling sites	# Taxa lists	Sampling period
Borbecker Mühlenbach	1	1	2012
Boye	3	9	2004-2013
Deininghauser Bach	13	93	1996-2012
Dellwiger Bach	5	44	1994-2012
Dorneburger Mühlenbach	7	23	2002-2012
Emscher	3	3	2012
Haarbach	2	4	2012-2013
Katzbach	1	11	1998-2012
Kirchschemmsbach	2	5	2009-2013
Laeppkes Mühlenbach	5	35	1996-2012
Nattbach	1	2	2012-2013
Vorthbach	3	14	1999-2013
Wittringer Bach	2	4	2012-2013

Environmental parameters

For each sample 87 environmental parameters (Appendix 1, Table A1) were collected. These environmental parameters were either taken directly from the monitoring protocols and additional information from the EGLV, or have been specifically calculated by GIS-analyses.

Eleven environmental parameters refer to the sample: age of restoration (years), meaning time since restoration at time of sampling, presence of iron ochre (yes/no), the share of microhabitats on the river bed (stones and gravel, sand and clay, artificial substrates, algae and macrophytes and living parts of terrestrial plants, organic matter, dead wood); or to the site: length of the restored section (m), occurrence of sewage overflows upstream of a sampling site (yes/no), and connection to a near-natural upstream section or tributary (yes/no). Three environmental parameters concern the physical habitat quality (PHQ) of the sampling sites and the section upstream of the sampling sites; for this purpose the data collected by LANUV NRW were used (source: Landesamt für Natur, Umwelt und Verbraucherschutz © Land NRW, Recklinghausen, http://www.lanuv.nrw.de). The evaluation of the physical habitat quality was calculated for 100 m sections and is based on a seven-point scale (1 = very good; 7 = bad, LUA NRW 1998; Gellert et al. 2014). For the final hydromorphological score of a stream section information about watercourse development, longitudinal profile, bottom structure, transverse profile, riparian structure, and the streams' adjacent areas are taken into account (LUA NRW 1998). In our analysis the hydromorphological score of the sampling site and the mean score of the sections 200 m upstream and 1000 m upstream were considered.

Data of the Official Topographical-Cartographic Information System (ATKIS®, spatial resolution 3 m) was used to evaluate the share of land use in the sub-catchment and in riparian buffers of different length and width. Prior to the analysis, the ATKIS® land use types were summarised to seven categories: extensively built-up settlement area (gardens, parks, cemeteries, etc.) intensively built-up settlement area (= sealed area), agricultural land, grassland, deciduous and mixed riparian vegetation, coniferous riparian vegetation and surface waters.

Based on the results of Kail & Hering (2009), the buffer lengths were set to 100 m, 200 m, 500 m and 1000 m, with widths of 20 m and 100 m. Thus, eight different buffers were produced with different width to length combinations. All tributaries within the requested buffer length were included. Finally, the share of land use and the share of areas with sealed surface were calculated within these buffer areas and sub-catchments. In addition, the share of contaminated areas within the sub-catchments was derived from GIS-data of the EGLV.

The water chemistry was only indirectly included in the analyses due to missing or invalid one-time sampling data, but land use and contaminated areas were regarded as proxies for water chemistry.

2.2.3 Assessing the influence of environmental parameters on the Ecological Potential

To analyse the influence of environmental parameters on the Ecological Potential non-metric multidimensional scaling (NMS) with the assemblages of all samples was carried out. NMS was based on the Bray-Curtis similarity index with log x+1 transformed abundance data. In the NMS plots we highlighted the Ecological Potential, the age of restoration, the river type and the sampling method to check if these parameters coincide with community composition and if the sampling method and/or the river type interfere with the analysis.

In a further step, the shares of samples meeting the water management target (Ecological Potential of "very good" or "good") and not meeting the target (Ecological Potential of "moderate", "poor", and "bad") were calculated for sites with different environmental characteristics. For the environmental parameters that are not "yes/no" encoded (as e.g. "share of unsealed area in the sub-catchment"), the mean of all samples of the respective environmental parameter was selected as the boundary. An exception is the environmental parameter "age of restoration". Here, the boundary was set at 9 years after restoration following the results of a previous study (Winking et al. 2014).

Principal Component(s) Analyses (PCA) were applied to elaborate the relation of the Ecological Potential to the environmental parameters and to rate the relative importance of environmental parameters for the Ecological Potential. The PCA were performed using CANOCO 4.5 (ter Braak & Smilauer, 2003). To ensure comparability of the length of the vectors a "mean = (- variance one)" standardisation was performed. A logarithm of the data for adaptation to a normal distribution was not performed, because outliers have already been deleted at the beginning of the analysis. For scaling, the distance between the samples was used ("inter-sample distances"). The relevant axes were identified for each data set by regarding the highest eigenvalues.

The first PCA (eigenvalues: 0.596-0.183) included all 87 environmental parameters and all samples (n = 248). Environmental parameters with a statistically low influence (vector length < 0.2) on the Ecological Potential were further on excluded from the analysis (Appendix 1, Table A1). We retained only a single parameter of closely related environmental parameters (e.g. a certain land use types in buffer zones), even if several of these were influential on the Ecological Potential; in these cases, only the parameter with the greatest impact was used for further analysis (Appendix 1, Table A1). After this reduction, 11 of 87 environmental parameters remained in the analysis. In total, three PCA were calculated: The first with all samples, the second only with samples taken with the Perlodes-method (n = 92) to test for potential methodological biases and to reduce the dominance of samples from a single stream, the Deininghauser Bach (from 93 to 15 samples). The third PCA included only the samples of sites restored at least 9 years ago (n = 128) to test if the influence of restoration age is diminishing 9 years after restoration (according to the results of Winking et al. 2014).

2.3 Results

2.3.1 Influence of the environmental parameters on the Ecological Potential

About 44 % of all 248 samples indicate a "good" or "very good" Ecological Potential (Figure 2.2). 17 of the 48 sampling sites (35.4 %) have already reached the Good Ecological Potential at the most recent sampling (Figure 2.3). On the basis of time series, the development of the assessment of each site was compared (Table 2.2). Some sites reach the Good Ecological Potential already shortly after restoration (Bor1, Boy1, Dor1), while the Ecological Potential of many sites improved over the time (e.g. Dei3, Dei9-11, Boy1, Vor1, Vor3, Läp3), decreased (Boy3, Vor2, Dor4) or remained constant over time (e.g. Dei6, Del3, Dor3).
The assemblages of sampling sites with a Good Ecological Potential differed significantly from the assemblages with a "poor" or "bad" Ecological Potential (ANOSIM at $R > 0.5$ and a $p < 5$ %; Figure 2.4(1)). Although the sampling sites at which the Good Ecological Potential was reached were mainly located in stream sections that have been restored 9 years or more years ago (Figure 2.4(1) and (2)), no significant difference was found between sites that have been restored more or less than 9 years before (ANOSIM at $R = 0.26$ and a $p < 5$ %). There was no significant difference in Ecological Potential between stream types and sampling method (Figure 2.4(3) and (4)).

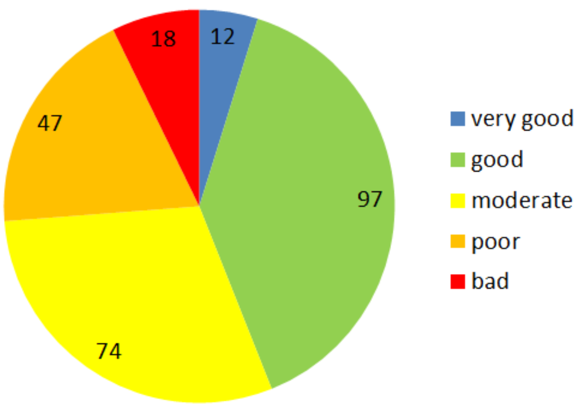

Figure 2.2 Ecological Potential of all samples (n = 248) assessed by HMWB-Type: "Flood Protection and Urbanisation (with foreland)".

The share of samples meeting the WFD target (Ecological Potential of “very good” and “good”) compared to the share of samples not meeting this target (Ecological Potential of "moderate" or worse) revealed the role of the environmental parameters for the achievement of the Good Ecological Potential (Figure 2.5). The Good Ecological Potential was mainly observed in stream sections with the following characteristics: high share of unsealed surface in riparian buffers and sub-catchments upstream of the sampling site, high share of deciduous woody riparian vegetation upstream of the sampling site, restoration performed at least 9 years before, connectivity to a near-natural upstream section, physical habitat quality class at the sampling site of "3" or better and dead wood present in the river bed (Figure 2.5). All other environmental parameters did not affect the Ecological Potential.

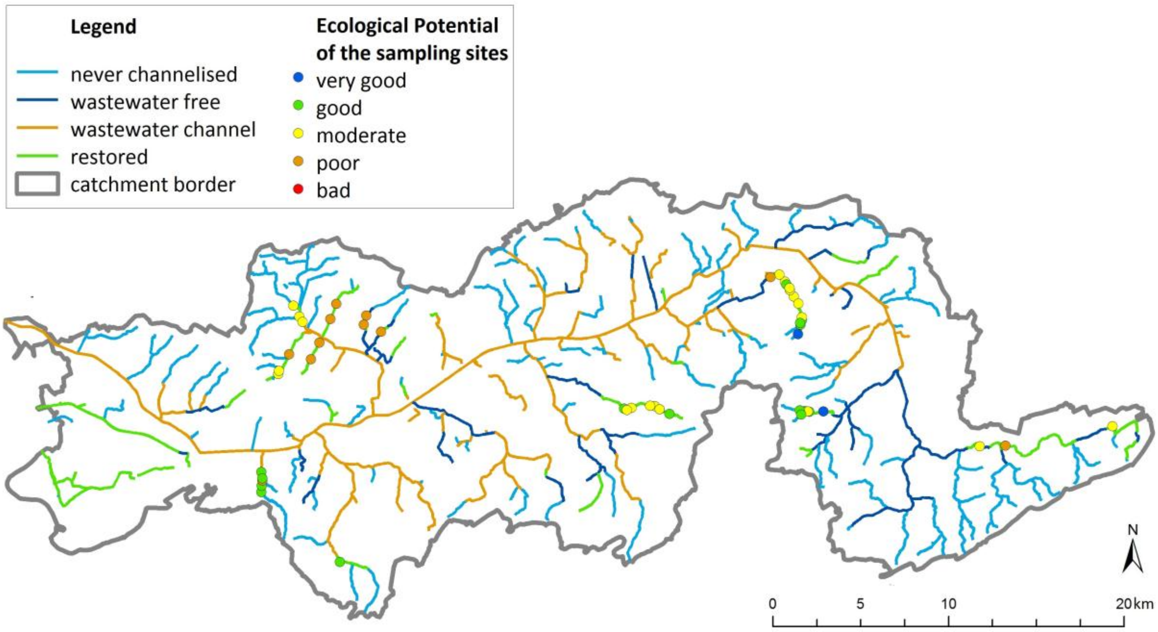

Figure 2.3 Ecological Potential of the sites at the youngest sampling event.

The PCA calculated with all 248 samples revealed the “age of restoration” as the most important parameter influencing the Ecological Potential, followed by “unsealed area in the buffer of 100 x 1000 m”, while the “presence of iron ochre” was negatively correlated (Figure 2.6). The results of the PCA solely calculated with the samples that were taken by the Perlodes-method (n = 92) also indicate the “age of restoration” as most influential. Other parameters such as “presence of dead wood”, “unsealed area in the buffer of 100 x 1000 m”, “deciduous riparian vegetation in the buffer of 20 x 1000 m” (positively) and “presence of iron ochre” (negatively) also affected the Ecological Potential (Figure 2.7). In both PCA the effects of sewage overflows and of unsealed area in the sub-catchment were comparatively small.

The PCA calculated with the samples taken 9 years or more years after restoration (n = 128), identified the “occurrence of sewage overflows upstream the sampling site” as the most influential parameter (Figure 2.8), followed by “unsealed area in the buffer of 100 x 1000 m” and “share of deciduous riparian vegetation in the buffer of 20 x 1000 m”. The “share of macrophytes” was identified to have a negative influence on the Ecological Potential. The influence of the “age of restoration” on the Ecological Potential was lower compared to the two other PCA.

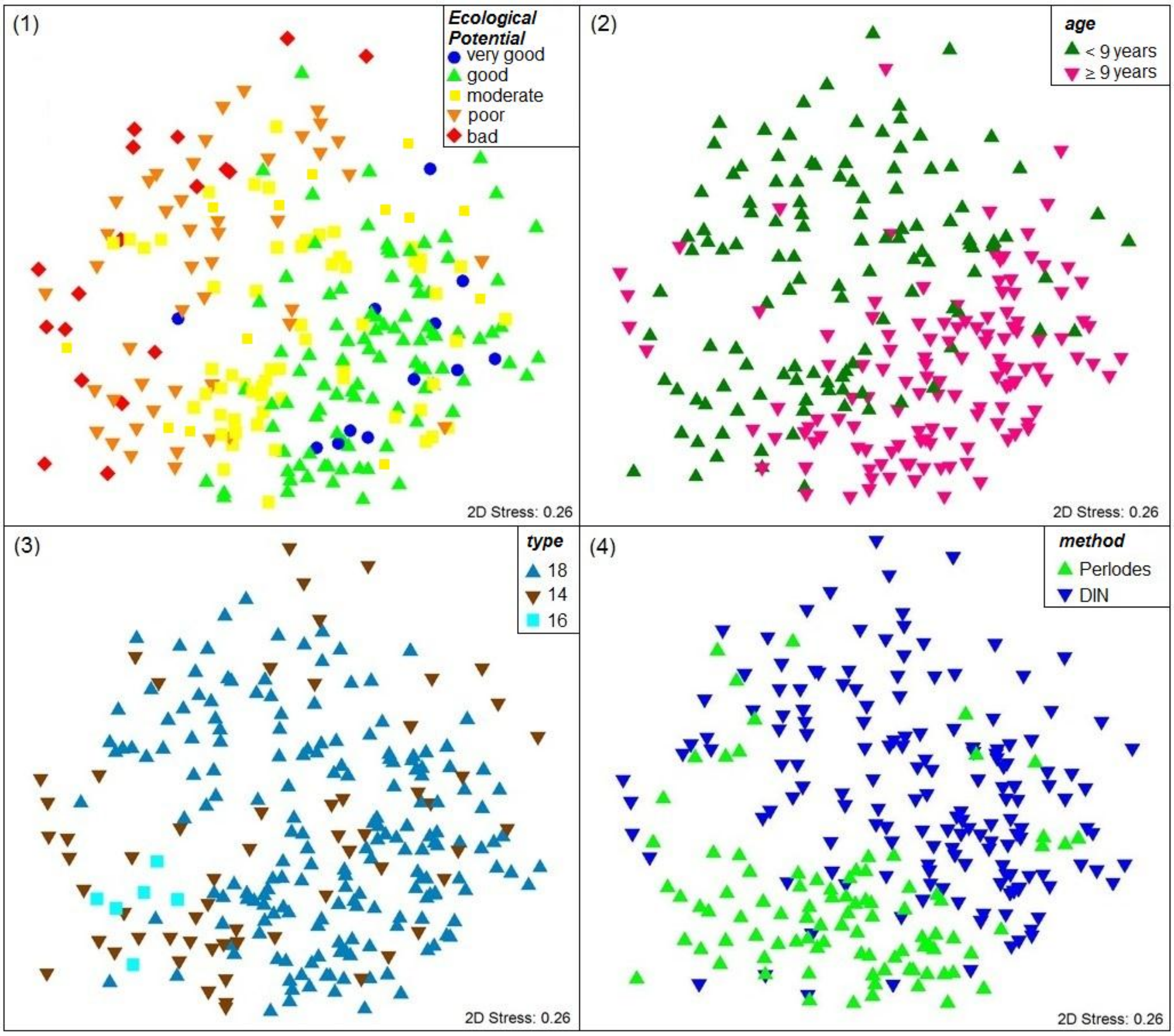

Figure 2.4 Non-metric multidimensional scaling (NMS) of 248 benthic invertebrate assemblages highlighted according to (1) Ecological Potential, (2) age of restoration, (3) stream type and (4) sampling method: Perlodes = multi-habitat-sampling according to Meier et al. (2006), DIN = DIN 38410.

Table 2.2 The Ecological Potential of the sampling sites in time series and additional information of the sites´ sub-catchment size and stream type. The numbers behind the sites names represent the longitudinal location at 1 = site nearest to source. The assessment of the Ecological Potential from 1 = very good to 5 = bad correspond to the mentioned colour coding in Fig. 2.2) Only one sampling event of a year was considered (respectively the earliest year) for this time series representation. Thus, 219 taxa lists were used.

Water body	Site	Sub-catchment size (km²)	Stream type	Age of restoration (years)																									
				1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	27
Borbecker Muehlenbach	Bor1	8.23	18	2																									
Boye	Boy1	15.92	14	4				3																					
	Boy2	17.48	14									3	3																
	Boy3	24.71	14	1			2			2		2	3																
Deininghauser Bach	Dei1	0.90	18												1														
	Dei2	1.87	18		2	2	2	3	2	2	2	1	2	3	2	2	2	2	2		2								
	Dei3	1.94	18				4	2	3	2	2	3		3	2														
	Dei4	2.05	18					2																					
	Dei5	2.11	18								3	3	2	3															
	Dei6	2.60	18	3	3	3	3	3		3																			
	Dei7	2.95	18				3	2	3	3																			
	Dei8	3.30	18				3	4	3	3																			
	Dei9	3.38	18	4	4	5	4	4		4	3																		
	Dei10	3.46	18								3	4	2		2														
	Dei11	3.75	18	4	4	5	4	3	4	2	4	3	3	3	4		3												
	Dei12	5.80	18					3		4		3																	
	Dei13	5.99	18	5	5				4																				
Dellwiger Bach	Del1	1.01	18														2	3	1	3	2	2							2
	Del2	1.26	18													1	2	2						2	2	2	2		
	Del3	3.18	18																3	3	3	3							
	Del4	3.22	18									4				2	2	2	1	3	2	2		2	2	1	2	2	
	Del5	4.08	18															2	3	2	2	2		2	2	2	1	2	1
Dorneburger Muehlenbach	Dor1	2.37	18	2				2						2															
	Dor2	2.83	18	3																									
	Dor3	3.12	18		3			2		3		3																	
	Dor4	3.50	18			2		3		3																			
	Dor5	5.71	18					3																					
	Dor6	5.98	18	3				3																					
	Dor7	6.40	18	4						3		4		2															
Emscher	Ems1	7.26	18	3																									
	Ems2	28.44	18		4																								
	Ems3	42.72	18		3																								
Haarbach	Haa1	4.73	14	3	4																								
	Haa2	7.40	14	5	4																								
Katzbach	Kat1	1.04	18												2	2	2	1	2					3	3	2	4	2	2
Kirchschemmsbach	Kir1	2.94	14				3	4																					
	Kir2	3.62	14	4			5	4																					
Laeppkes Muehlenbach	Läp1	3.49	18							1		2	2	2	2					2	2								
	Läp2	4.16	14							2	2	3	2	2	2	3	2			2				2					
	Läp3	4.23	14					2					5		5	4	4												
	Läp4	4.44	14							5	2	3	2	3	2		3				2			2					
	Läp5	4.50	14														2												
Nattbach	Nat1	3.28	14	5	4																								
Vorthbach	Vor1	2.27	14						3	5				4								3	3						
	Vor2	2.55	14						5	4		1		2								3	3						
	Vor3	6.74	16	5	4																								
Wittringer Bach	Wit1	3.37	16	4	4																								
	Wit2	3.89	16	4	4																								

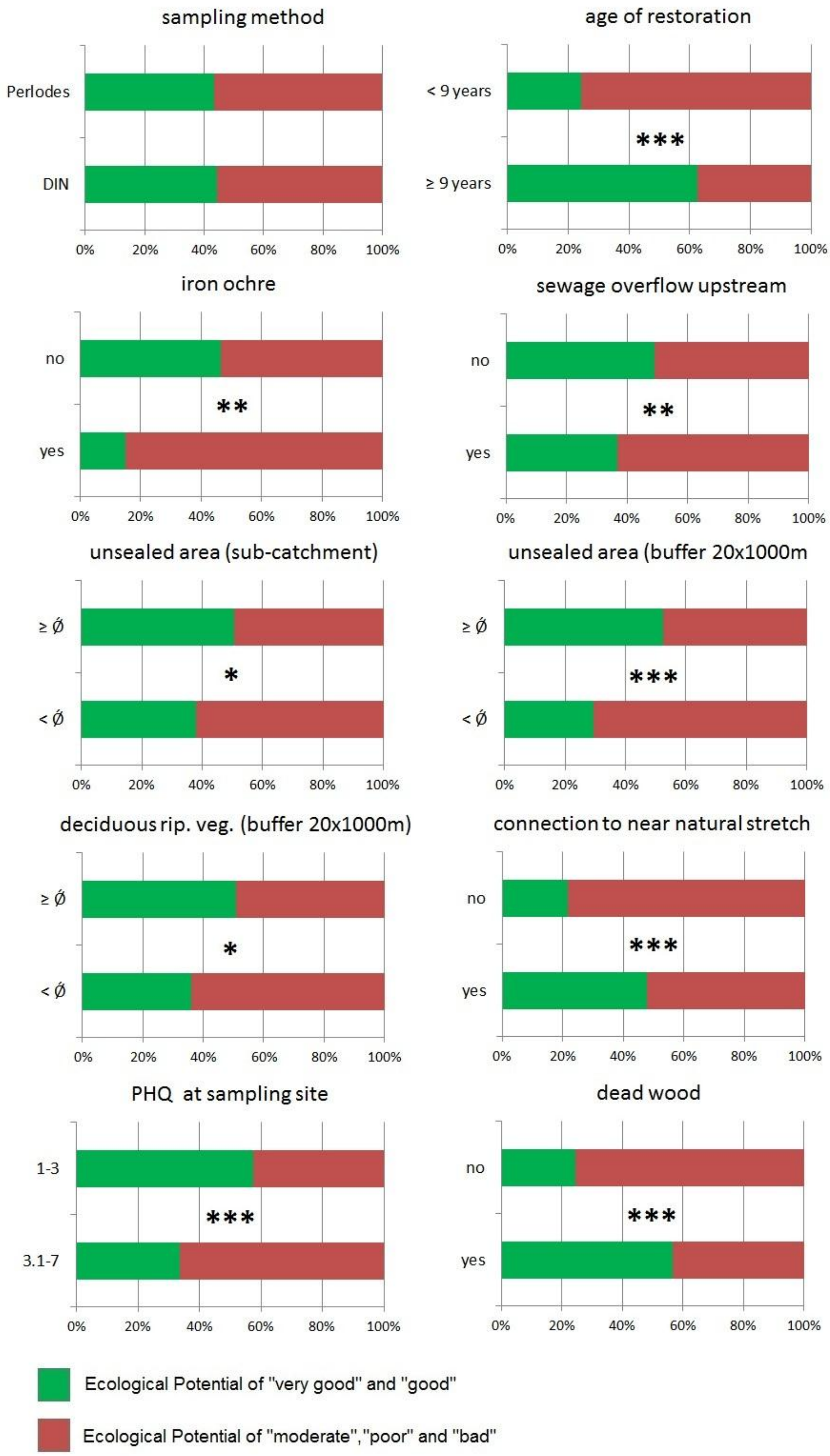

Figure 2.5 The share of samples meeting the WFD target (Ecological Potential "very good" and "good") as opposed to samples not meeting the WFD target (Ecological Potential "moderate", "poor" and "bad") under different environmental conditions. Perlodes = multi-habitat-sampling according to Meier et al. (2006), DIN = DIN 38410. rip.veg. = riparian vegetation.

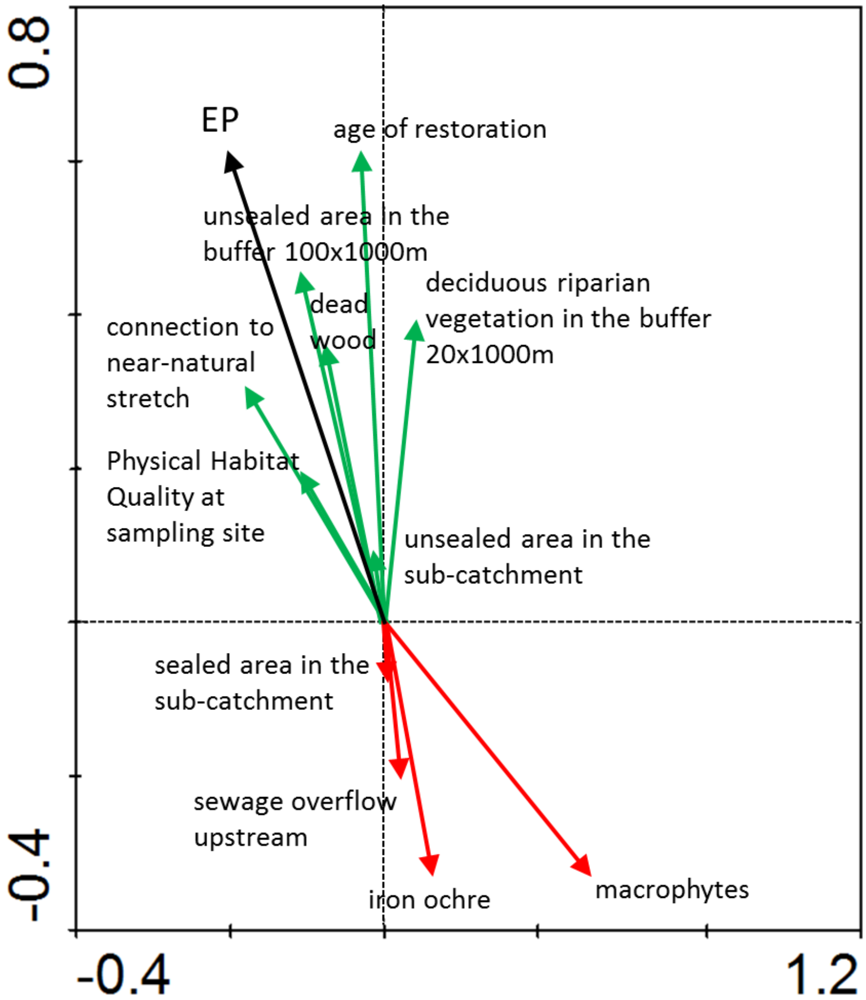

Figure 2.6 PCA of environmental parameters and the Ecological Potential (EP) calculated with all samples (n = 248).

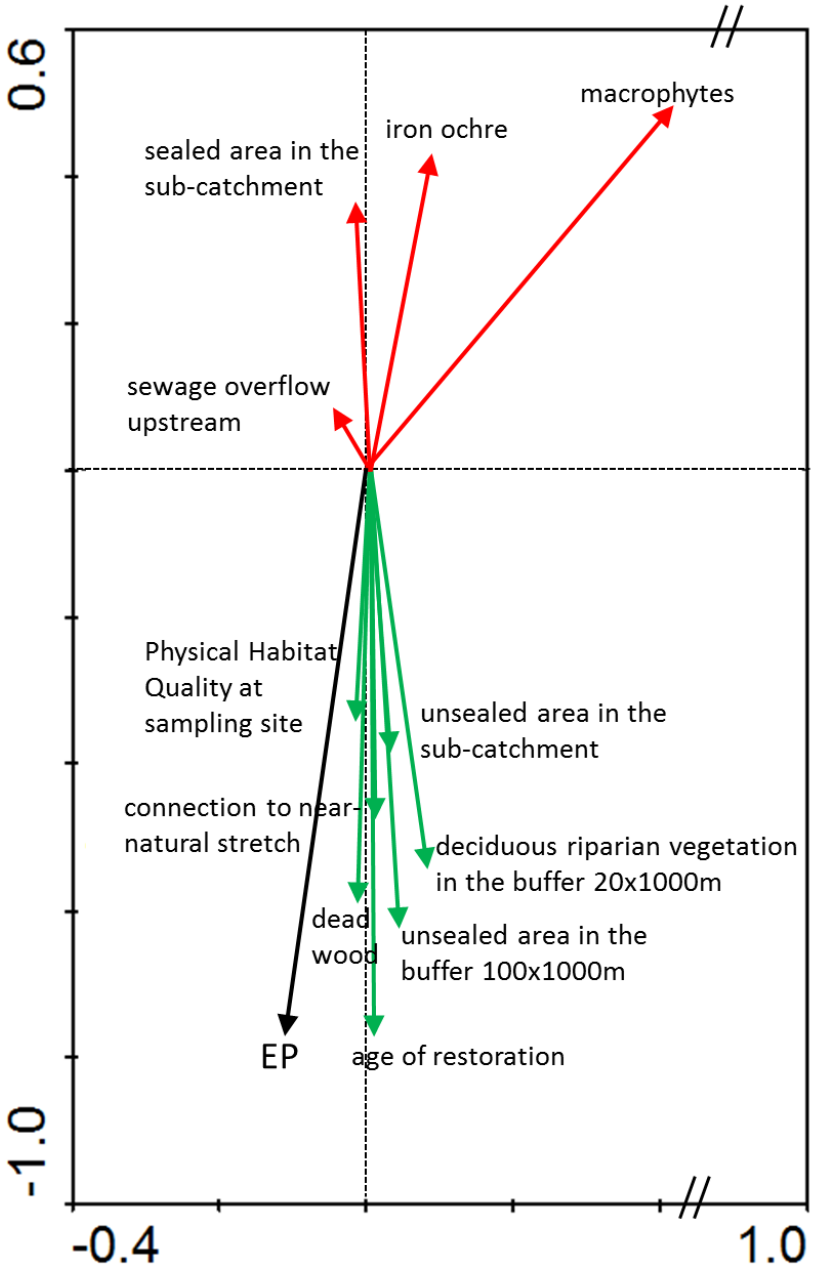

Figure 2.7 PCA calculated with PERLODES samples (n = 92). EP = Ecological Potential.

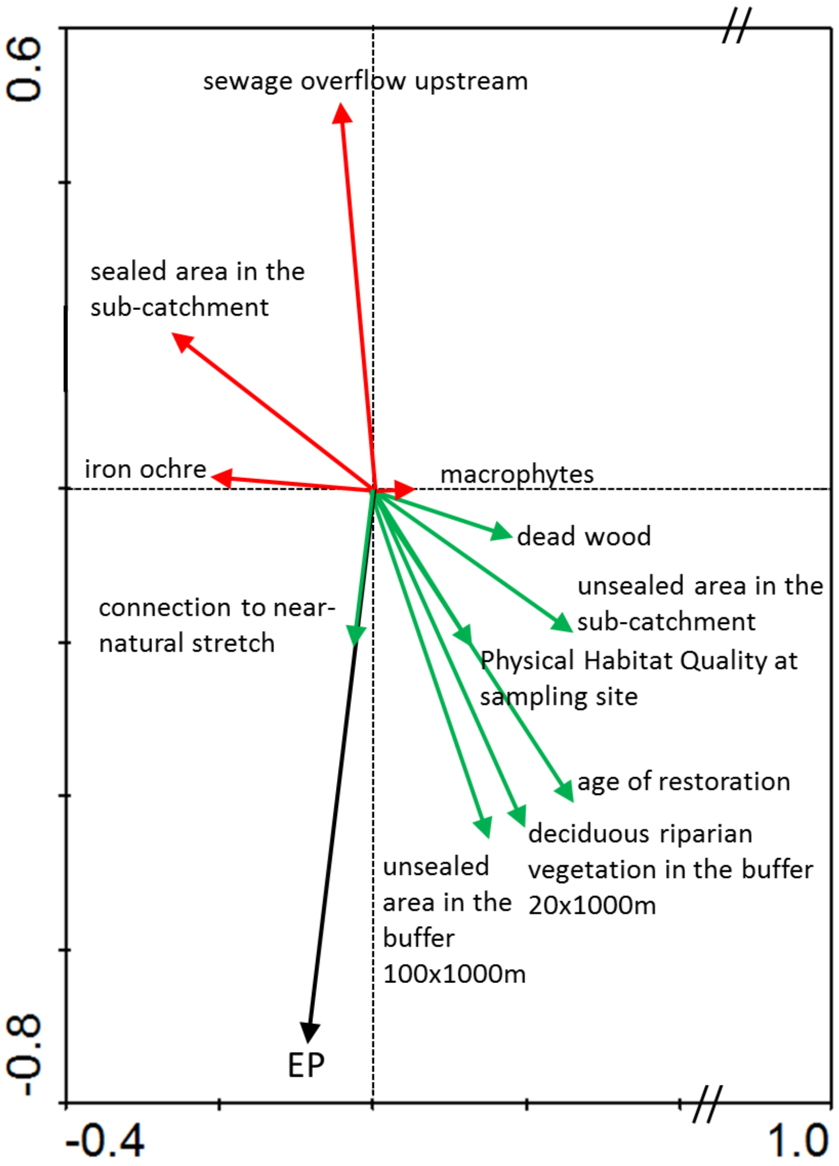

Figure 2.8 PCA calculated with samples taken 9 years or later after restoration (n = 128). EP = Ecological Potential.

2.4 Discussion

2.4.1 Environmental parameters affecting the Ecological Potential

The most influencing factor for the achievement of the Good Ecological Potential in streams of the Emscher catchment seems to be the age of restoration. Unlike restored streams in the open landscape, recolonisation of the virgin streams in the Emscher catchment has to start from scratch. The recolonisation by sensitive taxa, which are key indicators for a Good Ecological Potential (Hering et al. 2010), is related to their dispersal capability, to the recolonisation sources, the in-stream and riparian habitats of the restored streams and the habitat quality at the catchment scale (e.g. Sundermann et al. 2011; Tonkin et al. 2014; Jähnig et al. 2009 and Palmer et al. 2010), whereas the last parameters need time to develop (Kail & Hering 2005; Bell et al. 2013).

The development of the assemblages and the Ecological Potential of the individual sites vary greatly. In part, adjacent sampling sites differ significantly within a water body. This could have many reasons: the environmental conditions differ with sampling site and may change between nearby stream sections, e.g. in shading, impact of a sewage overflow or intense land use. This may explain why assemblages of some sites develop faster than others. Persisting water quality problems or missing habitat structures may explain why some sites have not yet reached the Good Ecological Potential, even after several years, or have even deteriorated. Hence, it is not just a matter of implementing a matter and having patience to reach the Good Ecological Potential. The age of restoration, which was the dominant factor for reaching the Good Ecological Potential, is a spurious correlation and can be interpreted as a proxy for the degree of the habitat development of the stream. This is underlined by the fact that three environmental parameters, which show a positive development of habitats (deciduous riparian vegetation, dead wood, and the Physical Habitat Quality), are positively correlated with the age of restoration. The presence of sewage overflows, which can hinder a positive development of in-stream habitats, is negatively correlated with the age of restoration. Maturation of the communities and attaining the Good Ecological Potential requires shorter time spans, if positively influencing environmental parameters are present shortly after restoration. The Good Ecological Potential can then be reached after 1 year or 2 years after restoration (Bor1, Dei2, Dor1), but is also observed to need about a decade (e.g. Dei9-10, Dor7). Both is surprisingly short for these virgin streams, as Jones and Schmitz (2009) found mean recovery times in restored freshwaters of 10 to 20 years, while Parkyn & Smith (2011) proposed 10-50 years for the maturation of restored sites if nearby recolonisation sources are present. This fits to our results, as deciduous riparian vegetation, a good hydro-morphological quality and dead wood on the streambed indicate the degree of maturation of the restored site.

The connection to a near-natural section also plays a key role for the recolonisation with sensitive taxa, hololimnic taxa and taxa with a low dispersal capability (Parkyn & Smith 2011; Winking et al. 2014). This connection is particularly important for the primary colonisation by drifting species (Gellert et al. 2012). Later, when riparian and in-stream habitats have developed, a connection to such a section is less relevant. But nevertheless, older restored sections still benefit from such a connection, because some sensitive taxa seem to need long time spans to establish populations or are dependent on mature habitats.

The positive effect of unsealed area in the surroundings of the restored streams was also evident. Unsealed soil acts as a buffer for inputs of matters and reduces diffuse pollution. A high share of unsealed area in the surroundings (buffer of 20 x 1000 m, or 100 x 1000 m above the sampling site) has thereby a stronger effect on the Ecological Potential than the unsealed area in the sub-catchment; a result that was also found in the study of Lorenz & Feld (2013).

In contrast, iron ochre on the stream bottom affects the Ecological Potential negatively. The clogging of substrates and the bonding of gills affect many macroinvertebrate species (Prange 2005). Sewage overflows that temporarily impair water quality primarily influence the assemblage of older restored sections, where riparian vegetation and in-stream habitats have developed and sensitive taxa have already recolonised. A strong determination of sewage overflows on benthic invertebrate communities and especially on sensitive species was also observed by Borja et al. (2006).

Surprisingly, the presence of macrophytes was negatively correlated with the Ecological Potential. Macrophytes provide habitat and food for other aquatic organisms, increase the structure- and flow diversity, contribute to the bottom and bank stabilisation and have diverse effects on the water chemistry, such as absorption of nutrients and heavy metals, and delivery of oxygen (Gregg & Rose 1982; Carpenter & Lodge 1986). In our results, the negative relation to the Ecological Potential probably reflects the age of restoration, hence the maturation of the section: young, nutrient-rich, still unshaded sections have optimal conditions for macrophytes. This can also be underlined by our results, as the share of macrophytes is negatively correlated with the age of restoration (PCA with all samples and with the samples taken by the Perlodes-method), while in the PCA restricted to samples in streams restored at least 9 years ago no correlation between the Ecological Potential and macrophytes was found.

2.4.2 Prognoses for future restoration projects

Some environmental parameters are mandatory for a rapid recolonisation success. Thus, the following recommendations can be derived from our results to optimise future restoration projects: creation/enhancement of growth of deciduous woody riparian vegetation along buffer strips of the streams, reduction and improvement of sewage overflows, provisioning of a connection to the streams tributaries, and active addition of dead wood to the streams.

The results are best transferable within the Emscher system and to other streams of the HMBW-type "Flood Protection and Urbanisation (with foreland)". Future restoration projects in streams with environmental parameters as the streams meeting a Good Ecological Potential have likewise chances to achieve the Good Ecological Potential. Orientation values, as means of the samples (n = 248) which already achieved the Good Ecological Potential, are:

- Minimum share of deciduous riparian vegetation in the buffer of 20 x 1000 m: 59.7 %
- Minimum share of deciduous riparian vegetation in the buffer of 20 x 500 m: 57.6 %
- Minimum share of unsealed area in the sub-catchment: 60.4 %
- Minimum share unsealed area in the buffer 100 x 1000m: 77.9 %
- Minimum share unsealed area in the buffer 20 x 1000m: 84.1 %
- Deadwood present on the stream bottom
- Connection to a near-natural tributary or upstream section
- Iron ochre absent

The results provide no indication of the amount of dead wood, which is necessary in order to achieve the Good Ecological Potential. As only values of 0%, 5% or 10% were observed, the data basis for a valid derivation of orientation values is not sufficient.

Our results show that even in a densely populated area with a complex pollution history many stream sections have good prospects of achieving the Good Ecological Potential - even if they predominantly transported waste water for 100 years.

3 Recolonisation patterns of benthic invertebrates: a field investigation of restored former sewage channels

3.1 Introduction

Numerous recent studies have analysed the response of benthic invertebrate assemblages to stream habitat restoration. While species number or diversity increased in some restored streams (Jungwirth et al. 1993; Gerhard & Reich 2000), the vast majority of restoration measures did not result in measurable changes to the benthic invertebrate assemblage (e.g. Feld et al. 2011; Friberg et al. 1998; Haase et al. 2013; Jähnig et al. 2010; Palmer et al. 2010; Sundermann et al. 2011b). The limited effects of restoration on benthic invertebrates are discussed in the above references. For example, benthic assemblages are strongly determined by factors acting at large spatial scales, ranging from water quality impacts from diffuse nutrient inputs to meta-population dynamics at the catchment or sub-catchment scales, which are not influenced by restoration occurring at the site scale.

The importance of the regional species pool as a perquisite for the recolonisation of restored sites by sensitive species is gaining increasing attention (Hughes 2007; Jähnig et al. 2010; Lake et al. 2007; Shields et al. 1995; Spänhoff & Arle 2007). Langford et al. (2009) suggest a direct correlation between recolonisation success and the distance to recolonisation sources. Empirical evidence, however, is scarce. Based on a data set of 24 restored sites, Sundermann et al. (2011a) documented a positive effect of source communities at distances of up to 5 km from restored sites.

In addition to the distribution of the regional species pool, time delays in the recolonisation of restored sites may be caused by species' dispersal capabilities and barriers between the restored site and potential recolonisation sources. The dispersal of most freshwater invertebrate species includes both active and passive components such as aquatic active (swimming, walking, etc.), aquatic passive (drift), aerial active (compensation flight or undirected flight of adults), aerial passive (wind drift, thermal drift as adults) and attachment to animals (e.g. to waterfowls, deer) (Bilton et al. 2001; Tachet et al. 2010; Van Leeuwen et al. 2013). While restored sites downstream from near-natural stream sections might easily be colonised via drift, weirs and culverts can limit recolonisation. Additionally, aerial dispersal of adult aquatic insects is not solely a question of distance but also of riparian land use, which may either enhance or inhibit dispersal (Lorenz & Feld 2013; Pander & Geist 2013).

Hence, studies of recolonisation of restored stream sections should consider the succession of the assemblage in the restored site, the local species pool, the dispersal mechanisms of the species involved and the barriers obstructing migration. In this context, the analysis of recolonisation patterns is almost always subject to two methodological problems: (1) source communities in the surrounding areas might have been overlooked and (2) species recorded in the restored sites and designated as successful colonisers might have been present prior to restoration.

Here, we analyse recolonisation patterns in restored stream sections in the Emscher catchment/Boye sub-catchment (Ruhr Metropolitan Area, western Germany), which are not affected by the second methodological problem above, as they have not previously been inhabited by any invertebrate taxa except Oligochaeta. Until recently, most streams in the Emscher catchment were concrete channels that exclusively transported sewage. As part of a large-scale restoration project, streams in the catchment are now being restored by constructing underground culverts for the sewage, followed by reconstruction of the channel bed and the riparian environment. All invertebrate species recorded in the restored areas are active or passive colonisers that reached the sections after restoration. By sampling several restored streams and nearby potential recolonisation sources, we addressed the following three hypotheses:

(H1) Restored sections undergo a succession from initial benthic invertebrate assemblages to more mature communities over several years.

(H2) Restored stream sections are first colonised by winged species and generalists, followed by more specialised winged species and later by wingless species.

(H3) The rate of recolonisation depends on the restored section’s location; restored stream sections with a near-natural, connected upstream section are more rapidly colonised than isolated sites due to drift effects. Generally, the progress of recolonisation is facilitated by the presence of nearby recolonisation sources.

3.2 Methods

3.2.1 Study area

The Boye is located in the catchment of the Emscher, a right tributary of the River Rhine. It drains an area of 77 km²; the downstream sections are predominantly urbanised, while the upstream parts are sections agricultural but mainly forested. The Boye's river network includes 90 km of total stream length. Like most of the Emscher river network, the downstream parts of the Boye and its main tributaries were transformed into concrete channels in the early 20th century and used to transport domestic wastewater to the Emscher and, eventually, to a central purification plant close to the mouth of the Emscher. Prior to restoration, these open sewers in the Boye catchment had a total length of 30.5 km. They were not colonised by benthic invertebrates except for rare occurrences of Oligochaeta (own observations in not-yet-restored neighbouring channels).

Since 1993, these open sewers have been undergoing restoration, which includes the construction of underground sewers for the wastewater, the subsequent removal of the concrete bed and the construction of a near-natural river channel and riparian areas. Subsequently, the restored channels are colonised by pioneer invertebrate assemblages. To date, 14.5 km of the former sewers, which comprise sections of the Boye and six tributaries, have been restored, while 16 km has remained unchanged and will be restored in the coming years.

The upstream sections of the Boye and its tributaries (59.5 km in total) have never been used as open sewers and remain in near-natural conditions (EGLV 2015). Along with the neighbouring stream systems, these upstream sections are supposed to be the main sources of recolonisation for the restored sections of the Boye system.

We sampled 45 sites in the Boye catchment and its neighbouring catchments, which comprise four water body groups (Figure 3.2):

(1) Restored sections in the Boye catchment: formerly open sewers that were restored between 1993 and 2011 (seven sites in three streams) and are connected with an upstream section (restored connected sites, RC).

(2) Restored sections in the Boye catchment: former sewers that were restored between 2008 and 2011 (six sites in four streams) and are not connected with an upstream section (restored unconnected sites, RU).

(3) Upstream sections in the Boye catchment: these sections have never been used as open sewers, have always retained their benthic invertebrate assemblages and could act as

recolonisation sources for the restored sections through drift or aerial dispersal of invertebrates (21 sites in 14 streams) (source sites in the Boye catchment, SB).

(4) Sections in basins close to the Boye catchment: these sections have also always retained their benthic invertebrate assemblages and could potentially act as recolonisation sources for the restored sections in the Boye catchment through aerial dispersal of insects (eleven sites in eleven streams). All of these sites are less than 5 km from at least one of the restored sites (source sites outside of the Boye catchment, SO).

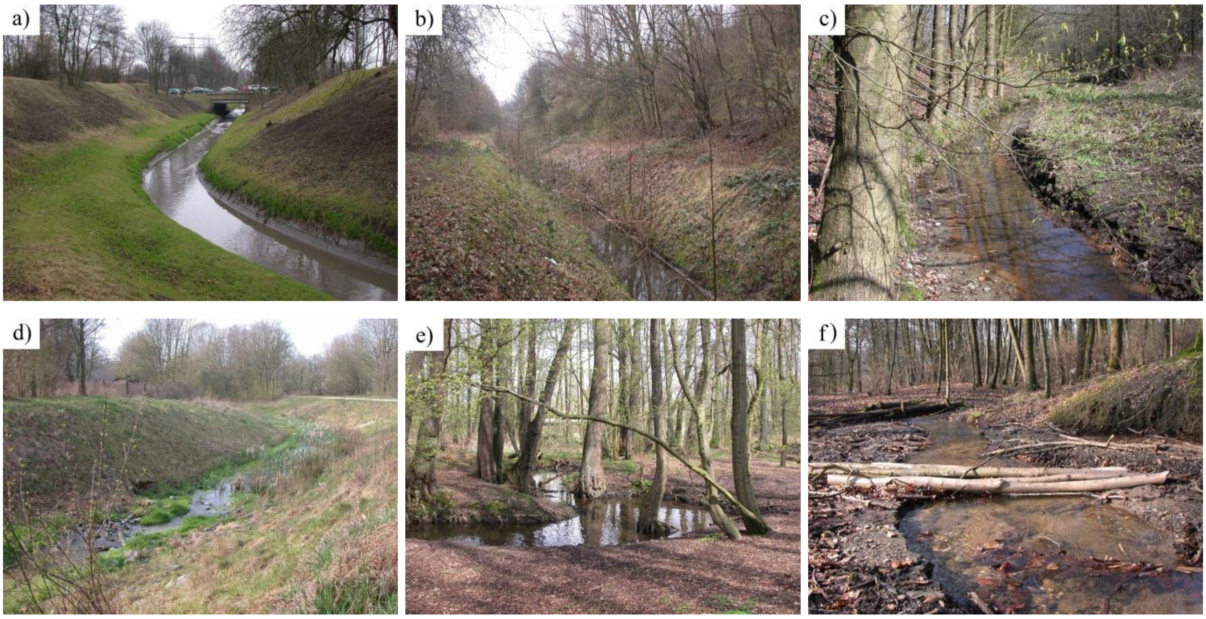

Figure 3.1 Example pictures of the streams prior to restoration and of the four water body groups. a) wastewater channel (Boye); b) channel formerly transporting wastewater (Wittringer Bach); c) connected restored section in the Boye catchment (RC): Vorthbach 19 years after restoration; d) unconnected restored section in the Boye catchment (RU): Haarbach, one year after restoration; e) Upstream section in the Boye catchment (SB): Spechtsbach; f) Upstream section close to the Boye catchment (SO): Alsbach.

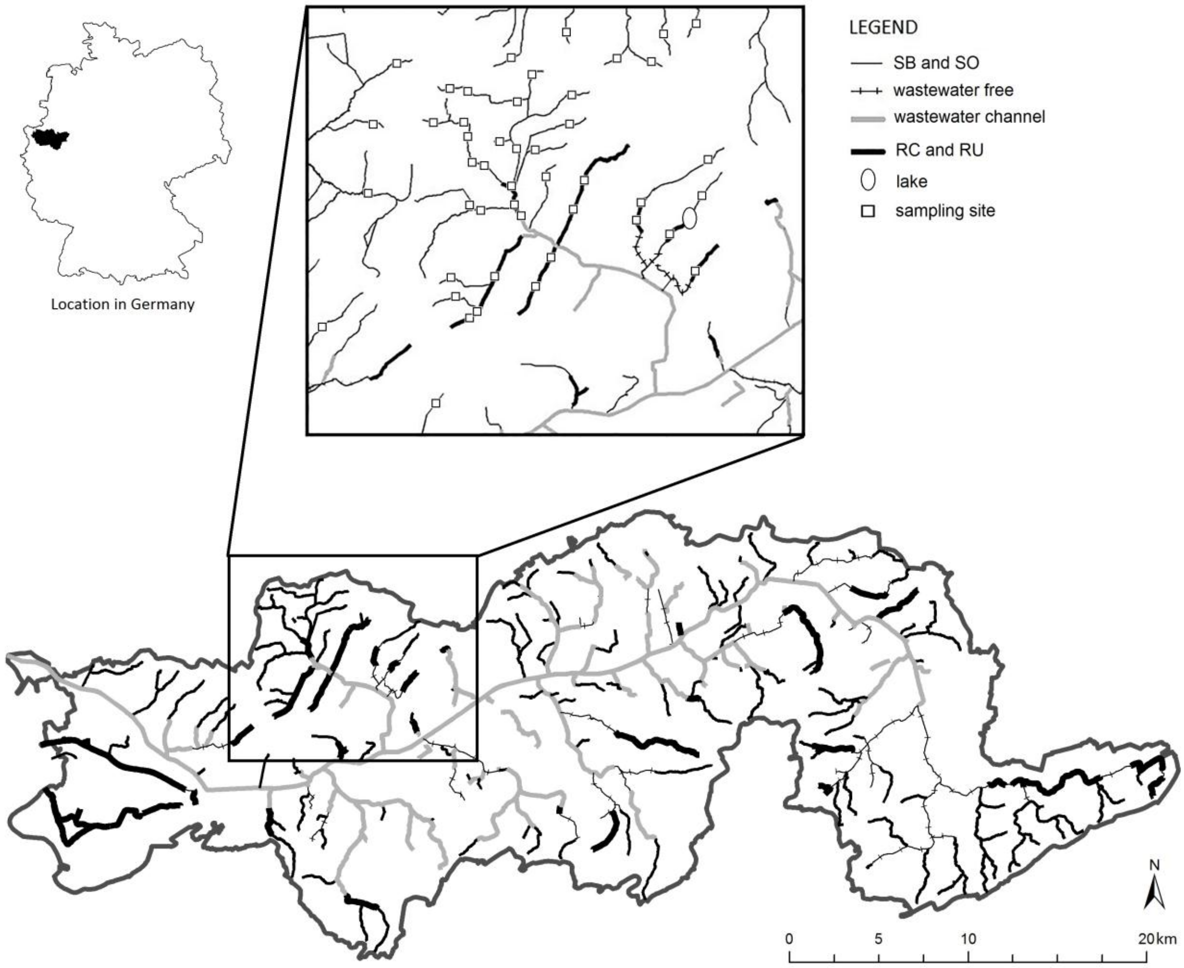

Figure 3.2 The study area and its location in the Emscher catchment area including the locations of the sampling sites. SB = source stream section within Boye catchment; SO = source stream section outside Boye catchment, RC = restored stream section connected to upstream near natural stream section; RU = restored stream section unconnected to any upstream near natural stream section.

The same restoration measures were performed in all of the restored sections, so the starting conditions are comparable. Underground wastewater sewers were constructed, bed and bank fixations were removed, the streambed was widened, and if possible dikes were removed. Following these measures, pioneer stages of floodplain forest were developed by natural succession. Commonalities of all restored sites are catchment areas of less than 24.7 km², channel width of less than 3 meters, and a distance to source of less than 5,169 m. However, the restored sites differ both in size (the Boye is a 3rd order stream, while the tributaries are mainly 1st order streams or in one case a 2nd order stream) and time since restoration (1-19 years) (Table 3.1). Following our hypotheses, we related the occurrence of individual invertebrate species and the taxonomic composition in restored sites to a) the distance from recolonisation sources, b) the age of the restored site and c) the species' dispersal capabilities and their degree of ecological specialisation.

3.2.2 Sampling, sorting and identification

The 45 sites were sampled once between mid-March and mid-April, 2012, during conditions of low to medium flow. Multi-habitat samples reflecting the proportion of the present microhabitat types were taken from each site (Haase et al. 2006). Each sample was comprised of 20 subsample units (25 x 25 cm) taken with a hand net (500 µ mesh size) from all microhabitats with > 5 % coverage, which resulted in 1.25 m^2 of the stream bottom being sampled. The 20 subsample units were pooled into one main sample, sorted in the field according to the German standard protocol (Meier et al. 2006) and preserved in ethanol (96 %). To ensure consistency in microhabitat estimation and invertebrate counting, all samples were taken and sorted by one person (myself). In the laboratory, individuals were identified to the level of species or genus (according to Haase et al. 2011), except for some Oligochaeta and most Diptera, which were identified to the subfamily or family level. The resulting taxa lists were adjusted prior to analysis (Nijboer & Schmidt-Kloiber 2004). The application of this standardised protocol and subsequent taxonomic adjustment ensures low variability in sampling, sorting and the resulting taxa list (Haase et al. 2004).

Table 3.1 Restored sites included into the sampling programme. Recolonisation source: upstream or downstream sections have never been used as open sewers. ds = downstream, us = upstream. Numbers in sampling site names refer to the location in the river course with 1 = site nearest to the source.

Water body name	Boye		Vorthbach			Wittringer Bach		Kirchschemmsbach		Haarbach		Nattbach	Hahnenbach
Sampling site	By1	By2	Vor1	Vor2	Vor3	Wit1	Wit2	Kir1	Kir2	Haa1	Haa2	Nat	Hah
Catchment size (km²)	24.7	17.5	2.3	2.6	6.7	3,4	3.9	3	3.6	4.7	7.4	3.3	0.7
Stream order	3	3	1	1	2	1	1	1	1	1	1	1	1
Distance to source	4775	5169	607	908	2048	3051	3588	1008	1675	1836	2810	2470	1026
Time since restoration (years)	9	9	19	19	1	1.5	1.5	4	4	1	1	1	1
Position of the closest recolonisation source	us	us	ds	ds	us	us	us	-	-	-	-	-	-
Distance to closest recolonisation source (m) (drift)	1301	881	400	111	229	591	1178	-	-	-	-	-	-

3.2.3 Dispersal classes

Based on an extensive survey of the literature, the sampled taxa were classified into five classes that reflected their dispersal capabilities:

Dispersal class A: Hololimnic, wingless taxa without the ability for aerial dispersal
Dispersal class B: Winged adult stage, low dispersal capabilities, habitat specialists
Dispersal class C: Winged adult stage, high dispersal capabilities, habitat specialists
Dispersal class D: Winged adult stage, low dispersal capabilities, habitat generalists
Dispersal class E: Winged adult stage, high dispersal capabilities, habitat generalists

The categorisation was mainly based on Bis & Usseglio-Polatera (2004), Schmedtje & Colling (1996), Schmidt-Kloiber & Hering (2012), Tachet et al. (2010), Vieira et al. (2006), and several papers addressing individual taxa (Appendix 2, Table A2).

Taxa with low dispersal capabilities were defined as those meeting one of the following criteria: (1) ability to fly less than 1 km before oviposition according to Vieira et al. (2006); (2) low dispersal capabilities according to taxa-specific references; (3) less than 40 % aerial dispersal according to Bis & Usseglio-Polatera (2004) and/or Tachet et al. (2010).

Taxa with high dispersal capabilities were defined as those meeting one of the following criteria: (1) ability to fly more than 1 km before oviposition according to Vieira et al. (2006); (2) middle or high dispersal capabilities according to taxa-specific references; (3) 40 % or greater aerial dispersal according to Bis & Usseglio-Polatera (2004) and/or Tachet et al. (2010). If plausible, the dispersal classification derived from Bis & Usseglio-Polatera (2004) and Tachet et al. (2010), which is given only for genus, was extended to species (Appendix 2, Table A2).

Habitat specialists were defined as those taxa that preferred (1) a single microhabitat according to Schmidt-Kloiber & Hering (2012), Schmedtje & Colling (1996) or taxon-specific references or (2) springs (crenal) (> 4 points according to Schmidt-Kloiber & Hering 2012). Preferences for macrophytes (phytal) or mud (pelal) were ignored as these habitats mainly occur dominantly in degraded areas.

Habitat generalists were defined as those taxa meeting one of the following criteria: (1) preference for several microhabitats according to Schmidt-Kloiber & Hering (2012), Schmedtje & Colling (1996) or taxon-specific references; (2) limnophilic taxa (> 4 points according to Schmidt-Kloiber & Hering 2012); (3) warm stenotherm or eurytherm according

to Schmidt-Kloiber & Hering (2012); (4) a wide range of longitudinal occurrence and possible occurrence in large rivers (> 4 points according to Schmidt-Kloiber & Hering 2012).
Four taxa (*Pilaria* sp., *Tipula* sp., *Simulium* sp. and *Prodiamesa olivacea*) poorly represented in the literature were re-assigned to dispersal classes based on the sampling results (Figure 3.3).

3.2.4 Comparison of sites

We calculated the Jaccard (presence/absence) and Bray-Curtis (abundance) similarity indices between the assemblages at all sites. Average similarities were calculated for pairs of restored sites, pairs of source sites and pairs of restored and source sites.
Non-metric multidimensional scaling (NMS) was applied to the assemblages at all sites. NMS was based on the Jaccard similarity index and the Bray-Curtis similarity index (log x+1 transformed), thus regarding presence/absence of taxa as a proxy for successful dispersal and abundance of taxa as a proxy for population establishment. One outlier site (the same one) was excluded from each NMS analysis. Analysis of similarity (ANOSIM) was applied to test for differences between the assemblages of the water body groups (RC, RU, SB, SO) and between young restored sites (1-4 years) and old restored sites (9-19 years). ANOSIM tests if taxonomic composition of sample groups is significantly different. The difference (R, range: -1 to +1) between averages of ranked similarities was used to test if the similarity is higher within one group or between groups. The higher the value of R, the better the separation between groups (R > 0.75: well separated; R > 0.5: overlapping but clearly different; R ≤ 0.5: barely separable; R = 0: no difference between groups). A p-value of < 5 % indicates a significant result (Clarke 1993).
The four water body groups were further compared using the Bray-Curtis similarity index (log x+1 transformed) (SIMPER; similarity percentages) and taxa number and abundance (Mann-Whitney U-test).

3.2.5 Comparison of taxa

The occurrences of each taxon in the 45 sampling sites were plotted on maps (example maps s. Figure 3.3) to provide an overview of the patterns of recolonisation. For each taxon, we further calculated the recolonisation quotient as:

$$RQ = \frac{frequency\_in\_restored\_sites\_[\%]}{frequency\_in\_source\_sites\_[\%]}$$

The recolonisation quotient is a proxy for successful recolonisation dispersal, not for successful establishment. A high quotient suggests a taxon with high dispersal capability. The mean recolonisation quotient was calculated for each dispersal class.

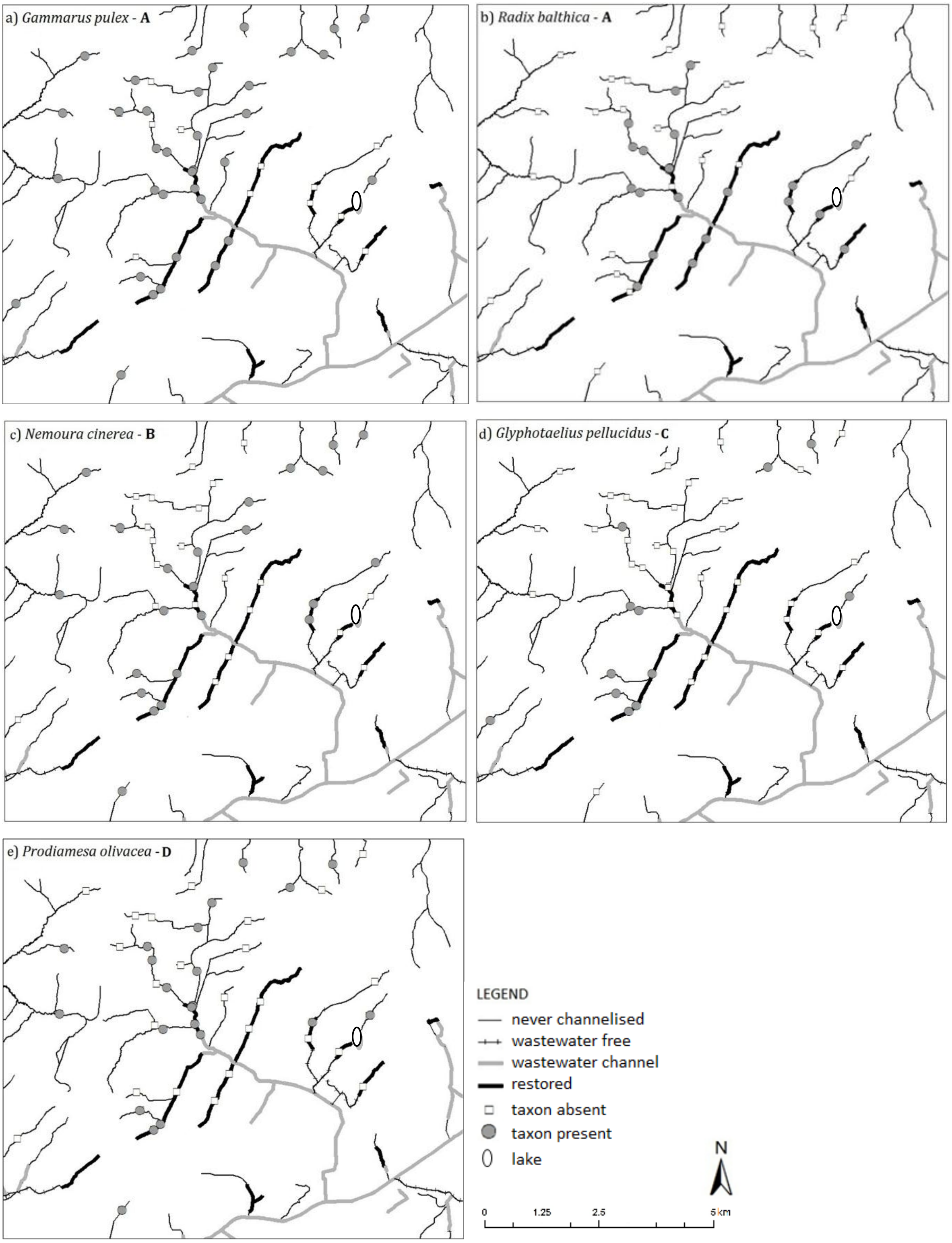

Figure 3.3 Dispersal maps of a) Gammarus pulex, b) Radix balthica (dispersal class A), c) Nemoura cinerea (dispersal class B), d) Glyphotaelius pellucidus (dispersal class C), e) Prodiamesa olivacea (dispersal class D), f) Pilaria sp. and g) Cloeon dipterum (dispersal class E).

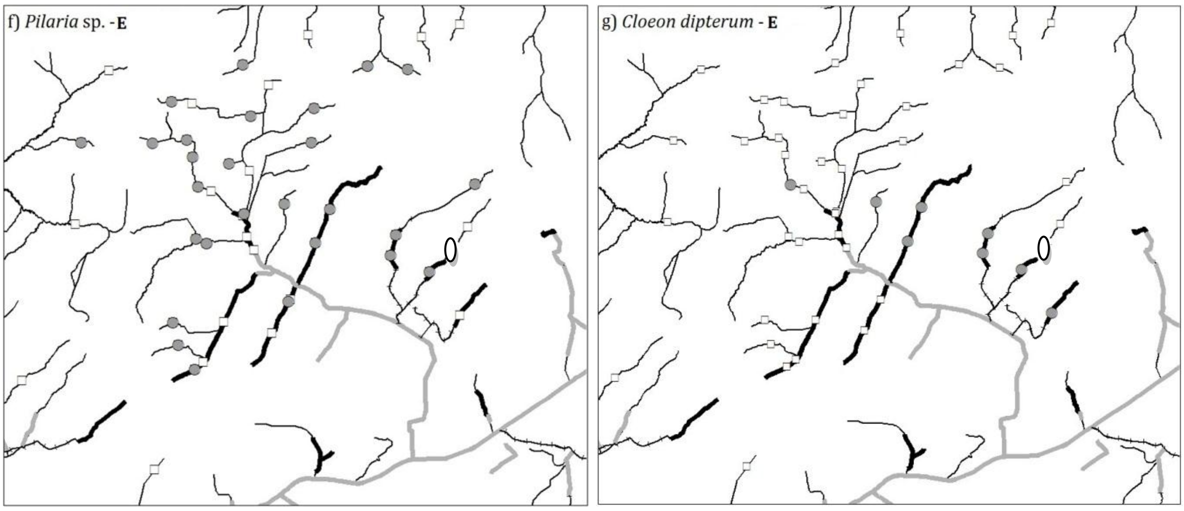

Figure 3.3 Continued.

3.2.6 Matching dispersal classes and the characteristics of restored sites

We related the composition of dispersal classes per restored sampling site to connectivity (sites with and without source sites upstream), distance to an upstream source section and time since restoration. The numbers of taxa representing the different dispersal classes were first compared between connected and unconnected sites (Mann-Whitney U-test). For the connected sites, we compared two groups of sites representing different time spans since restoration: young, connected (1-1.5 years) and old, connected (9-19 years) sites. We further related the number of taxa in each dispersal class to the distance to the nearest upstream source (linear regression).

3.2.7 Dispersal pathways

For all of the restored sites and the observed taxa, we measured the distance to the closest source community in GIS/ArcMap 10 (ESRI, Redlands, USA). If the closest source community was located upstream, drift distance was measured. If the closest source community was located in an adjacent stream, the aerial distance was measured. The possible dispersal pathways for recolonisation from the closest source sites to the restored sites were categorised as follows: a) drift, b) aerial dispersal from sources within the Boye catchment, c) aerial dispersal from sources outside of the Boye catchment but within a 5 km radius of the restored sites and d) aerial dispersal from sources outside the Boye catchment and more than 5 km distant. Drift (a) was always given priority as we considered drift to be the most likely pathway, even if recolonisation sources requiring aerial dispersal were closer. We then averaged the dispersal pathways for each of the dispersal classes A-E and calculated their relative importance.

We compared the likely recolonisation pathways for invertebrates between young, connected (1-1.5 years) and old, connected (9-19 years) restoration sites. This analysis was limited to taxa in the B and C dispersal classes as only the occurrence of these “habitat sensitive taxa” differed between “young” and “old” sites.

3.3 Results

3.3.1 Overall assemblage structure

Altogether, 128 taxa were recorded, of which 15 were found exclusively in restored sites, 62 in both restored and source sites, and 51 exclusively in source sites. On average, 20 taxa were found at restored connected (RC) sites (n = 7; range: 17 to 31) with a mean abundance of 1,263 ind. m^{-2} (range: 610 to 4,128). At the restored unconnected (RU) sites (n = 6), 17 taxa were recorded on average (range: 11 to 36) with a mean abundance of 962 ind. m^{-2} (range: 580 to 1,443). Number of taxa was not significantly different between RC and RU sites. At the source sites, 113 taxa were recorded. At source sites in the Boye catchment (SB sites, n = 21), 21 taxa were found on average (range: 14 to 29) with a mean abundance of 1,582 ind. m^{-2} (range: 344 to 8,915). At the source sites outside of the Boye catchment (SO sites, n = 11), 18 taxa were recorded on average (range: 9 to 24) with a mean abundance of 765 ind. m^{-2} (range: 193 to 1,874).

Qualitative assemblage similarity within source sites was 24.6 % (Jaccard index average) and quantitative assemblage similarity was 31.9 % (Bray-Curtis index average). The assemblage similarity values within restored sites were 25.3 % (Jaccard index average) and 36.1 % (Bray-Curtis index average), while assemblages in restored and source sites were less similar (19.7 % Jaccard index average; 27.8 % Bray-Curtis index average). In general, restored and source sites were well separated in both NMS plots (Figure 3.4), but the assemblages of the older (9-19 years) connected restored sites (Boye and Vorthbach) were similar to those of the source sites in the Boye catchment. The assemblages of the young restored sites (1-4 years) were strongly separated from the source sites. The taxonomic compositions of restored unconnected sites (RU) differed significantly from the source sites (SB and SO) (ANOSIM based on the Jaccard index, $R > 0.5$, significance level ≤ 5 %; Table 3.2), whereas the restored connected sites (RC) did not differ from the source sites (ANOSIM based on both the Jaccard and the Bray-Curtis index, $R < 0.5$, significance level ≤ 5 %; Table 3.2).

Also the taxonomic compositions of “young” restored sites compared to the “old” restored sites differed significantly from each other. The taxonomic compositions did not differ significantly among the other water body groups (RC and RU, SB and SO) (ANOSIM based on the Jaccard index, Table 3.2). The lowest Bray-Curtis average similarity was found between RU sites and SB sites and between RU sites and SO sites (SIMPER, Table 3.2).

Restored and source sites differed in their composition of invertebrate dispersal classes (Table 3.3). In general, source sites are characterised by higher abundances of hololimnic taxa (dispersal class A), more taxa from and higher abundances of dispersal class B (winged, lowly dispersing habitat specialists) and more taxa from dispersal class D (winged, lowly dispersing habitat generalists). Restored sites are characterised by more taxa from and higher abundances of dispersal class E (winged, strongly dispersing habitat generalists).

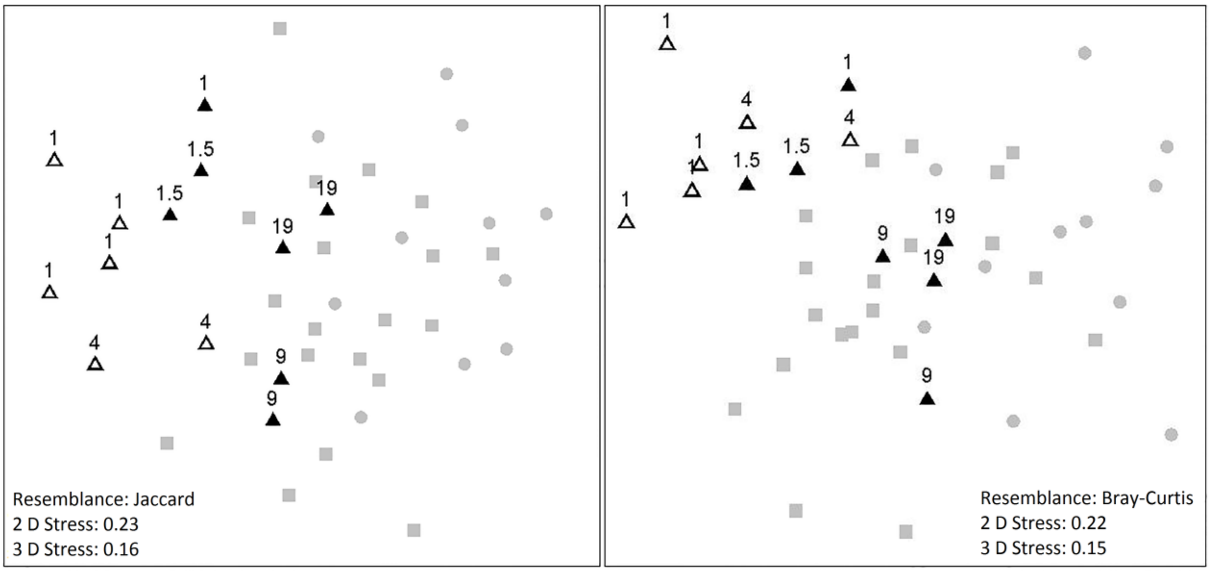

Figure 3.4 Non-metric multidimensional scaling (NMS) ordinations of benthic invertebrate assemblages in sampling sites of the four water body groups (n = 44; Jaccard index and Bray-Curtis index): full triangles = restored, connected site (RC); open triangles = restored, unconnected site (RU); grey rectangle = source within Boye catchment (SB); grey circle = source outside Boye catchment (SO). Numbers = years since restoration.

Table 3.2 Pairwise assemblage comparison of the water body groups and of old and young restored sites (ANOSIM based on Jaccard and Bray-Curtis index) and pairwise Bray-Curtis-average similarity (SIMPER). If R-value equals 1: similarity within the groups is higher than similarity to sites of the other group (R > 0.75: well separated; R > 0.5: overlapping, but clearly different; R ≤ 0.5: barely separable). A p-value of < 5 % indicates a significant result (see methods chapter). For abbreviations of water body groups compare Figure 3.2.

	Jaccard		Bray-Curtis		
Groups	R-value	P-value [%]	R-value	P-value [%]	Average similarity [%]
(1) RC, (2) RU	0.381	1.8	0.397	0.1	32.30
(1) RC, (3) SB	0.174	4.7	0.119	12	34.67
(1) RC, (4) SO	0.323	0.5	0.36	0.1	28.05
(2) RU, (3) SB	0.527	0.1	0.477	0.2	26.81
(2) RU, (4) SO	0.805	0.1	0.752	0.1	16.72
(3) SB, (4) SO	0.128	4.1	0.269	0.1	29.85
(5) young, (6) old	0.618	0.1	0.569	0.1	30.00

Table 3.3 Comparison of dispersal class composition between restored (R) and source sites (S). Ø# = average number; ind. m⁻² = individuals per square meter; Sig = level of significance (= p ≥ 0.05 (Mann & Whitney 1947)). Dispersal classes: A = hololimnic taxa; B = low dispersal capabilities and habitat specialists; C = high dispersal capabilities and habitat specialists; D = low dispersal capabilities and habitat generalists; E = high dispersal capabilities and habitat generalists; x = without classification.*

	R (n=13)	S (n=32)		R (n=13)	S (n=32)	
Class	Ø#Taxa	Ø#Taxa	Sig	Ø# Ind. m^{-2}	Ø# Ind. m^{-2}	Sig
A	4.8	5.7		355	765	*
B	0.6	1.3	*	14	68	*
C	1.0	1.7		13	20	
D	1.9	3.1	*	186	80	
E	8.7	6.2	*	459	286	*
x	1.7	1.6		95	83	
Sum	18.7	19.6		1,122	1,302	

3.3.2 Matching dispersal classes and characteristics of restored sites

The representation of dispersal class was similar between connected and unconnected sites with dispersal class E (winged, strongly dispersing habitat generalists) being the most species rich, followed by dispersal classes A (hololimnic taxa), D (winged, lowly dispersing habitat generalists), C (winged, strongly dispersing habitat specialists) and B (winged, lowly dispersing habitat specialists). However, numbers of taxa in the dispersal classes differed between connected and unconnected sites (Table 3.4). In particular, habitat specialists (dispersal classes B and C) were virtually absent from unconnected sites (except for a single record of *Tinodes waeneri waeneri* in Haa1), while they occurred regularly, albeit in low species numbers, at connected sites. Generalist taxa representing dispersal class E, however, were significantly more abundant at unconnected sites (Table 3.5).

As connectivity strongly determines the composition of the dispersal classes, the roles of the factors “age” and “distance to recolonisation source” were only analysed for connected sites. Significantly more taxa representing dispersal class C were found at old, connected sites (9-19 years) compared to young, connected sites (1-1.5 years), while there was no difference between the other dispersal classes (Table 3.5). There was little or no influence of the distance to recolonisation source on dispersal class composition (linear regression; R^2 for all dispersal classes < 0.5).

Table 3.4 Number of taxa and number of taxa per dispersal class for restored sites. Abbreviation of sampling sites according to Table 3.1. Dispersal classes: A = hololimnic taxa; B = low dispersal capabilities and habitat specialists; C = high dispersal capabilities and habitat specialists; D = low dispersal capabilities and habitat generalists; E = high dispersal capabilities and habitat generalists; x = without classification.

	Sampling site	Number of taxa	Dispersal classes					
			A	B	C	D	E	x
Connected	By1	17	7	0	1	2	6	1
	By2	31	8	1	5	6	8	3
	Vor1	19	5	1	2	3	6	2
	Vor2	17	6	1	3	2	3	2
	Vor3	18	5	2	1	2	6	2
	Wit1	23	5	2	0	3	10	3
	Wit2	17	4	1	0	1	10	1
Unconnected	Kir1	19	6	0	0	3	8	2
	Kir2	26	7	0	0	2	15	2
	Haa1	15	0	0	1	0	13	1
	Haa2	18	4	0	0	1	12	1
	Nat	13	2	0	0	0	10	1
	Hah	11	3	0	0	0	7	1
	Average	18.8	4.8	0.6	1	1.9	8.8	1.7

*Table 3.5 Effects of connectivity to recolonisation sources (n = 13) and age (only connected sites; n = 7) of the restoration on number of taxa representing different dispersal classes. Sig = level of significance (Mann-Whitney-U-test): *significance determined ($p \leq 0.05$),** significance high ($p \leq 0.01$), ***significance very high ($p \leq 0.005$), p = significance value, DC = dispersal class: A = hololimnic taxa; B = low dispersal capabilities and habitat specialists; C = high dispersal capabilities and habitat specialists; D = low dispersal capabilities and habitat generalists; E = high dispersal capabilities and habitat generalists; n = number of taxa.*

	Connection					Age				
Sig		***	*		*			*		
p	0.130	0.004	0.04	0.06	0.03	0.07	0.076	0.048	0.354	0.138
DC	A	B	C	D	E	A	B	C	D	E
n	32	6	16	16	46	32	6	16	16	46

3.3.3 Dispersal pathways of taxa

We analysed the distribution patterns of habitat specialists (dispersal classes B and C), of which numbers of taxa differed between the “old” and “young” connected, restored sites. At young, connected restored sites (Vor3, Wit1 and Wit2), all of the habitat specialists were also found at the closest source sites. At old, connected restored sites (Vor1, Vor2, By1 und By2), habitat specialists (dispersal class C) were found that did not occur at the closest connected source sites, in particular *Anabolia nervosa*, *Potamophylax* sp. and *Potamophylax rotundipennis*.

The most likely dispersal pathways for all of the taxa recorded at the restored sites were categorised by measuring the distance to the closest recolonisation source and calculating the average distances (Figure 3.5). Taxa from dispersal class B exclusively occurred at restored sites that they could have reached via drift from source sites, whereas the taxa from the other dispersal classes predominantly depended on aerial dispersal. Particularly for taxa represented by dispersal classes A and E, the majority of communities in the restored sites likely originated from sources within the Boye catchment. The distances overcome to colonise restored sites averaged approximately 2,000 m (median) for taxa representing dispersal classes A, B, D and E and about 4,000 m for taxa representing dispersal class C (Figure 3.6).

The recolonisation success of taxa representing the different dispersal classes was also reflected by the mean recolonisation quotient: E (mean RQ: 1.01) → A (mean RQ: 0.33) → D (mean RQ: 0.32) → C (mean RQ: 0.2) → B (mean RQ: 0.07).

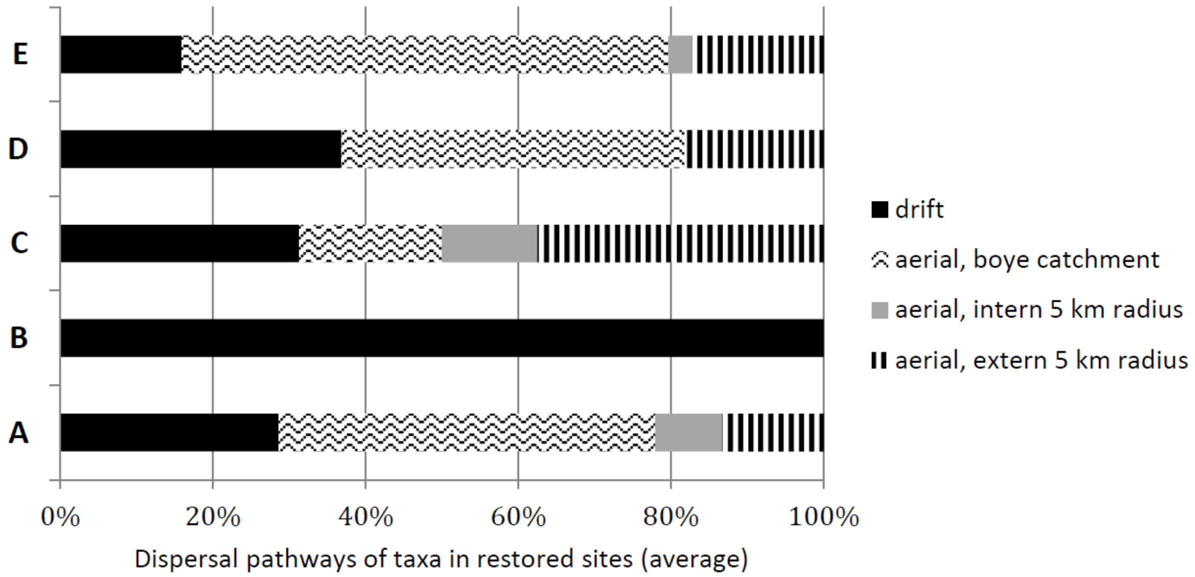

Figure 3.5 Most likely dispersal pathways for all taxa records in restored sites, broken down by the dispersal classes of the taxa. Dispersal classes: A = hololimnic taxa; B = low dispersal capabilities and habitat specialists; C = high dispersal capabilities and habitat specialists; D = low dispersal capabilities and habitat generalists; E = high dispersal capabilities and habitat generalists.

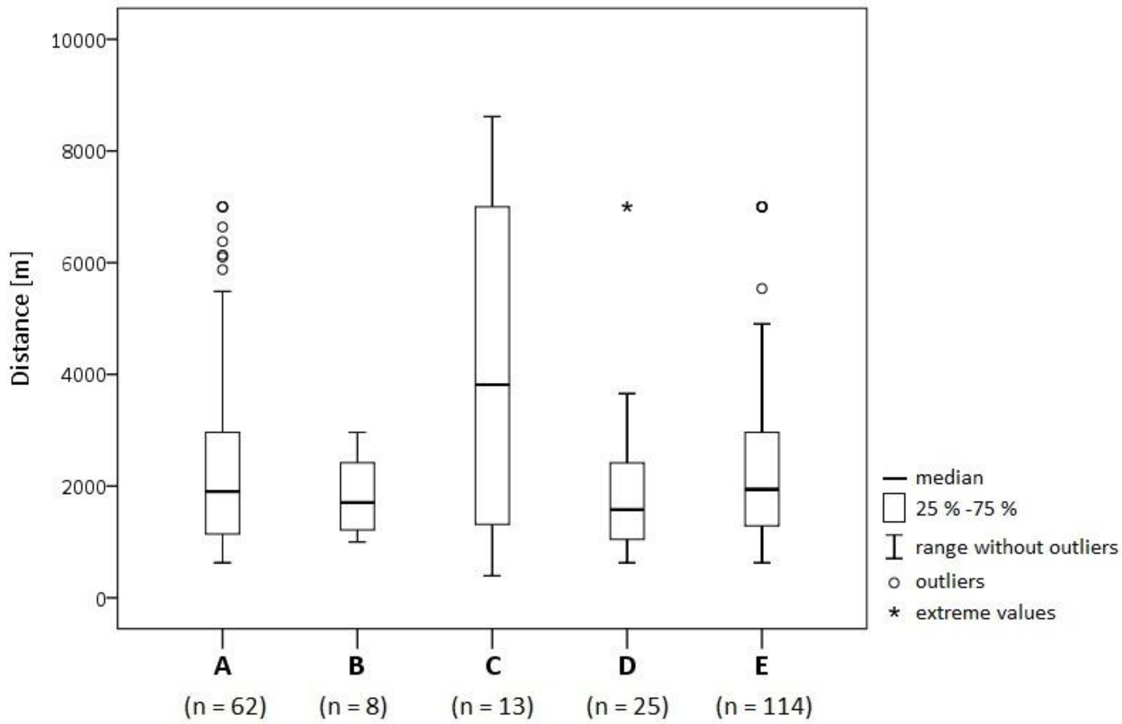

Figure 3.6 Distance to closest recolonisation source for all taxa records in restored sites, broken down by the dispersal classes of the taxa. Dispersal classes: A = hololimnic taxa; B = low dispersal capabilities and habitat specialists; C = high dispersal capabilities and habitat specialists; D = low dispersal capabilities and habitat generalists; E = high dispersal capabilities and habitat generalists.

3.4 Discussion

3.4.1 Succession

According to hypothesis H1, we expected the restored sections to undergo a succession from pioneer to more mature assemblages and this hypothesis was generally supported by our results. The assemblages of the old, restored sites resembled those of the source sites, while the taxonomic composition within the group of young, restored sites was less similar. Differences between the connected and unconnected sites were also apparent. The assemblages of the connected, restored sites matured after 9 to 19 years, while we expect succession to occur over longer time spans at the unconnected sites. The results of our study provide no evidence for the time required as the oldest unconnected site was restored 4 years prior to sampling and still has not matured. Overall, these time frames are comparable to those given by Becker & Robson (2009), who observed the maturation of benthic invertebrate assemblages 8 years after restoration of the riparian zone.

In terms of succession, a restored site may be defined as “mature” if the composition of the dispersal classes resembles that of the source sites. Particularly, mature assemblages in the Boye system are characterised by hololimnic species (dispersal class A) and by poorly dispersing winged species (dispersal classes B and D). This composition also indicates that habitats have developed, which is a requirement for specialists in dispersal classes B and C throughout their entire life cycle. For some taxa, e.g. *Glyphotaelius pellucidus*, woody riparian vegetation is necessary for oviposition. Shredders such as *Amphinemura* sp., *Halesus* sp., *Potamophylax* sp. or *Micropterna* sp. require leaves as a food source, and many Trichoptera need leaves for case building. The xylophagous caddis larvae, *Lype reducta*, needs dead wood as a food source. Such habitat specialists were not found in our young, unconnected sites but only in our source and old or connected sites. This is an indication that woody riparian vegetation supports the development of the invertebrate assemblage by providing habitats and food sources. Vegetation development requires time (e.g. Kail & Hering 2005; Bell et al. 2013), and vegetation-dependent recovery of freshwater systems is estimated to take one or two decades (Jones & Schmitz 2009), which is supported by our results.

Gustafsson et al. (2013) analysed the recolonisation of a newly built stream section by benthic invertebrates. After 2 years, approximately 60 % of the taxa present in the surroundings had colonised the section, whereas the missing 40 % were predominantly slow colonisers or taxa linked to riparian vegetation. In conclusion, the differences between the assemblages in our “young“ and “old“ sites may only partly be due to dispersal constraints but may also be determined by habitat availability.

3.4.2 Role of dispersal classes

We expected the restored sites to be first colonised by winged generalists and by hololimnic taxa last (hypothesis H2). This hypothesis was mainly rejected by our results. Source sites were generally characterised by hololimnic species, poorly dispersing winged specialists and generalists (dispersal classes A, B and D), which is in line with our succession related hypothesis. However, the composition of the dispersal classes in the restored sites and the “old” and “young” restored sites deviated considerably from our predictions. Dispersal classes colonised the restored sites in the following order: E, A, D, C, B.

The dominance of the taxa representing dispersal class E (winged, strongly dispersing generalists) was expected. In many cases, the closest recolonisation sources were several kilometres distant (e.g. the source for *Cloeon dipterum*, Figure 3.3). The combination of strong dispersal capability and low habitat specificity defines these taxa as pioneer species (sensu Gore 1982), which have a high probability of discovering restored sites and at the same time, the option to colonise sites with low habitat variability. Although Spänhoff & Arle (2007) expect recolonisation by aerial dispersal to often require several years, our results indicate that winged generalists are most effective at rapid, primary recolonisation.

The recolonisation success of the hololimnic species (dispersal class A) was, however, unexpected and might be explained as follows. First, many hololimnic species are highly abundant (such as Gammaridae, which has the overall highest abundance in the studied streams) and reproduce throughout the year (Hynes 1955). For example, several snails are capable of self-fertilisation and may spawn several times a year (Glöer & Meier-Brook 2003). The high population densities make dispersal by drift and subsequent establishment more likely. Drift is estimated to occur more often and faster than aerial dispersal (Spänhoff & Arle 2007). However, taxa in dispersal class A were also found in several of the unconnected sites, which they could not have reached by drift, and we did not observe a significant difference in the number and density of hololimnic species between the connected and unconnected restored sites. In 73 % of the cases, hololimnic species could not have reached the restored sites by drift. They may have had undetected communities in ditches, temporary water bodies or meadows in the floodplain (for lentic taxa) close to the restored sites and entered the stream when water levels were high (Langford et al. 2009). In exceptional cases, they may have survived in the former sewage channels. However, based on our own observations, the latter may only be true for some Oligochaeta and expected for taxa with high saprobic valences (e.g. *Physa fontinalis*, Saprobic Index of 3.6). Therefore, other dispersal pathways may be more important than initially expected, such as attachment to waterfowl or mammals (Van Leeuwen et al. 2013; Boulton et al. 1998) or anthropogenic influences. These dispersal modes appear to be effective but slower than the aerial dispersal of winged generalists (dispersal class E); only four hololimnic taxa were found in the 1 year old restored sites.

Winged generalists with low dispersal capability (dispersal class D) were almost exclusively present in connected or old unconnected sites. The low dispersal capability of these taxa is illustrated by the high proportion of sites that could have been reached by drift. The least successful colonisers were the taxa in dispersal classes B and C, which are characterised by specific habitat requirements. Lower habitat variability, such as the lack of particulate organic matter or dead wood in “young” restored sites, may have limited oviposition in dispersing adults or the survival of eggs and larvae. Particularly, the weakly dispersing habitat specialists (dispersal class B) were only found in connected restored sites, which likely receive a continuous inflow of specimens from upstream.

Sites were only sampled in the spring. Thus, some taxa, which might have successfully colonised restored sections, might have been missed by our sampling. However, as all sites were sampled during the same season, there is no bias in sampling design, and sampling in early spring allows for the detection of all hololimnic taxa and the majority of merolimnic taxa.

3.4.3 Role of the restored section´s location

We expected the recolonisation process to be accelerated due to nearby source communities (hypothesis H3), which was supported by our results.

In general, most of the taxa that recolonised the restored sites already had communities in the Boye catchment. Connected restored sites are inhabited by significantly more taxa in dispersal classes B and C than the unconnected sites. Additionally, in other studies, connection is the most important factor influencing the recolonisation of restored sites (e.g. Renöfält et al. 2005; Lake et al. 2007; Spänhoff & Arle 2007).

Apart from this general pattern, our study revealed the relevance of the species pool over greater distances. Fifteen taxa (12 % of all taxa) were recorded in restored sites but were not found in the source sites that were investigated (Appendix 2, Table A2). Therefore, unless they were overlooked, they must have dispersed more than 5 km to reach the restored sites. These taxa, such as *Anabolia nervosa*, *Potamophylax rotundipennis* or *Ischnura elegans*, are large species and good fliers and likely travelled the distance by flying. In other cases, the source communities in the surrounding of 5 km might have been overlooked. The majority of recolonisation events (88 %), however, resulted from source communities closer than 5 km, thus supporting the observations of Sundermann et al. (2011a), who also detected an impact from recolonisation sources within a 5 km radius.

4 Succession of benthic invertebrate assemblages in restored former sewage channels

4.1 Introduction

The restoration of rivers and streams results in drastic changes of the environmental conditions for benthic invertebrates: the water quality is improved, current patterns are changed and new habitats are generated. For the species present prior to restoration, these changes are disturbances, destroying or deteriorating their niches; at the same time, restoration creates niches for additional species or functional groups, which were rare or absent before (Connell 1978). The shift towards a post-restoration assemblage, however, is a process which requires time: new species need to move in and to establish populations, and these species might later on be replaced by others, which are more competitive (Connell 1978; McCook 1994). Further, the succession of the assemblage is shaped by stressors on both the local and the catchment scales. Some studies showed that catchment land use is an important factor for shaping benthic assemblages (e.g. Hughes et al. 2008), whereas other studies suggest that pressures on the reach scale are of similar importance (e.g. Verdonschot 2009). Thereunder, fine sediment entry (e.g. Von Bertrab et al. 2013), urban land use (e.g. Huang & Guo 2014), and missing recolonisation sources (Sundermann et al. 2011a; Tonkin et al. 2014) are expected to have a strong influence on the benthic invertebrates´ taxonomic composition in streams and therefore on the recolonisation process and succession.

For benthic invertebrates, one of the most commonly monitored organism group in streams, only little is known on the successional processes following restoration. The first assemblages establishing after disturbances are often described as pioneer communities consisting of well dispersing eurytopic taxa (Lorenz et al. 2009). Other studies described succession of benthic invertebrates in ponds (e.g. Boix et al. 2012; Ruhí et al. 2009; Miguel-Chinchilla et al. 2014), temporary wetlands (e.g. Boix et al. 2004; Ruhí et al. 2013) and lakes (e.g. Cañedo-Argüelles & Rieradevall 2011) and implied a rapid initial colonisation followed by a differentiation into habitat-specific assemblages and increasing taxa diversity.

One of the few well documented examples of invertebrate assemblage succession concerns the River Rhine in Germany, which was heavily polluted until the mid-seventies, before the water quality was gradually improved. The beginning ecological recovery was set back in 1986 when toxic fire runoff water ran into the river after an accident at the Sandoz chemical plant in Switzerland. Again, the macrozoobenthos community was eliminated over large sections of the upper Rhine. Subsequently, an increase in species richness, the establishment of various invasive species and a strong species turnover characterised by the almost complete replacement of invasive species by others took place. These observations, however, can not necessarily be transferred to other rivers or streams, as the Rhine is one the rivers most intensively used for navigation, thus favouring the establishment of invasive species (e.g. Nienhuis et al. 2002; Van den Brink et al. 1996). Furthermore, apart from the Sandoz accident, the changes in water quality were not drastic, but gradual and were superimposed by many other alterations, such as the Rhine-Main-Danube channel, which connected the Rhine catchment to the Danube and further increase the likelihood of invasive species establishment.

More typically, stream restoration creates new habitats in a short period of time, e.g. by remeandering. The effects of such measures on benthic invertebrates are in most cases minor (e.g. Feld et al. 2011; Jähnig et al. 2010; Palmer et al. 2010) and succession has almost never been documented. In conclusion, the understanding of succession processes in restored streams is currently limited.

The Boye catchment in western Germany offers perfect conditions for the analysis of succession processes, as the restored streams have been sewage channels for decades and were not inhabited by benthic invertebrates except Oligochaeta. Therefore, each taxon present in restored sites has colonised it after restoration, which included the improvement of the water quality and the near-natural construction of the stream bed and the riparian areas. In the previous study aiming at dispersal of taxa and the recolonisation of these former sewage channels, we observed a distinct recolonisation pattern (chapter 3). This pattern was based on the streams different ages of restoration and we found that winged, strongly dispersing generalists colonised restored sites most rapidly and were followed by hololimnic species, weakly dispersing generalists and habitat specialists.

In this paper we extended the data set used in chapter 3 and our research focus switched to the temporal process of recolonisation and to environmental parameters steering succession processes. Seven restored streams addressed in the previous study were re-investigated after 1 year. We tested the following two hypothesises:

(H4) Within 1 year time, the restored sites have been subject to further succession and maturation, which leads to a greater similarity of the assemblages to those of nearby source sites. Here, we expect a higher inter-annual change of young restored sites due to the rapid primary succession connected to a natural species turnover as described in many studies (e.g. Connell 1978; McCook 1994), and vanishing differences between assemblages of sites connected and unconnected to near-natural upstream reaches.

(H5) The state of maturation is determined by the environmental conditions of the restored sites. The succession towards near-natural assemblages is supported by recolonisation sources in the surroundings (i.e. the likelihood of dispersing specimens to reach the site, e.g. Sundermann et al. 2011a), a low share of urban land use in the surrounding (i.e. absence of barriers and pollutants, e.g. Huang & Guo 2014) and absence of sludge and sand on the river bottom (i.e. habitat quality; e.g. Von Bertrab et al. 2013).

4.2 Methods

4.2.1 Study area

We investigated the benthic invertebrate assemblages of 45 sites in the Boye catchment, which is part of the Emscher system in the western part of Germany. The Boye drains an area of 77 km² with a river network of 90 km stream length (Figure 4.1). The source of the Boye and the first tributaries are located in agricultural or forested areas, while the downstream parts and the downstream tributaries are mainly flowing through urban areas and public parks. Since the early 20th century, the downstream parts of the Boye and its tributaries have been used as open sewers, transporting domestic, untreated wastewater. These open sewers were constructed as concrete channels, which added up to a length of 30.5 km in the Boye catchment. The sewage allowed for no invertebrate assemblage besides Oligochaeta. Nowadays, 14.5 km of these former sewers, comprising parts of the Boye and six of its tributaries, have been restored. During the restoration process, which started in 1993, the concrete was removed, underground sewers for the wastewater were constructed and the river bed and the riparian areas were reconstructed to a near-natural state. The restored sections were subjected to colonisation by invertebrate assemblages from the surroundings (chapter 3). The remaining 16 km of open sewers are planned to be restored until the year 2020.

In contrast, the upstream sections of the Boye and its tributaries (59.5 km) have never been used as open sewers and have remained in a near-natural condition to date (EGLV 2015).

These upstream sections, along with near-natural sections of streams in neighbouring catchments, are the main sources of recolonisation for the restored sections in the Boye catchment (chapter 3; Winking et al. 2013).

Invertebrates were sampled in the Boye catchment and in neighbouring streams comprising the four water body groups (compare chapter 3, Figure 3.1):

(1) Restored sections in the Boye catchment: formerly open sewers which have been restored between 1993 and 2011 (seven sites in three streams) and are connected with an upstream section which has never been used as a sewage channel (RC).

(2) Restored sections in the Boye catchment: former sewers which have been restored between 2008 and 2011 (six sites in four streams) and are NOT connected with an upstream section which has never been used as a sewage channel (RU).

(3) Upstream sections in the Boye catchment; these sections have never been used as open sewers, always retained benthic invertebrate assemblages and could act as recolonisation sources for the restored sections through drift or aerial dispersal of invertebrates (21 sites in 14 streams) (SB).

(4) Sections in basins close to the Boye catchment; also these sections have always retained benthic invertebrate assemblages and could potentially act as colonisation sources for the restored sections in the Boye catchment through aerial dispersal of insects (eleven sites in eleven streams). All these sites are less than 5 km distant of the closest the restored sites (SO).

Comparable restoration measures, like the construction of underground wastewater sewers, removing of the concrete, stream widening and development of near-natural in-stream and riparian habitats, were performed in all restored reaches, thus leading to comparable starting conditions after restoration. Commonalities of all restored sites include a catchment area of less than 24.7 km², a channel width is of less than 3 meters, and a distance to source of less than 5,169 m. However, the restored sites differ in size: the Boye is a 3th order stream (2 sites), while the tributaries are 1st (10 sites) or 2nd (1 site) order streams; furthermore, they differ in connectivity and time since restoration. In-stream measurements and GIS analyses revealed differences in microhabitat composition, in the lands use and surface area of water bodies in the surrounding (chapter 3).

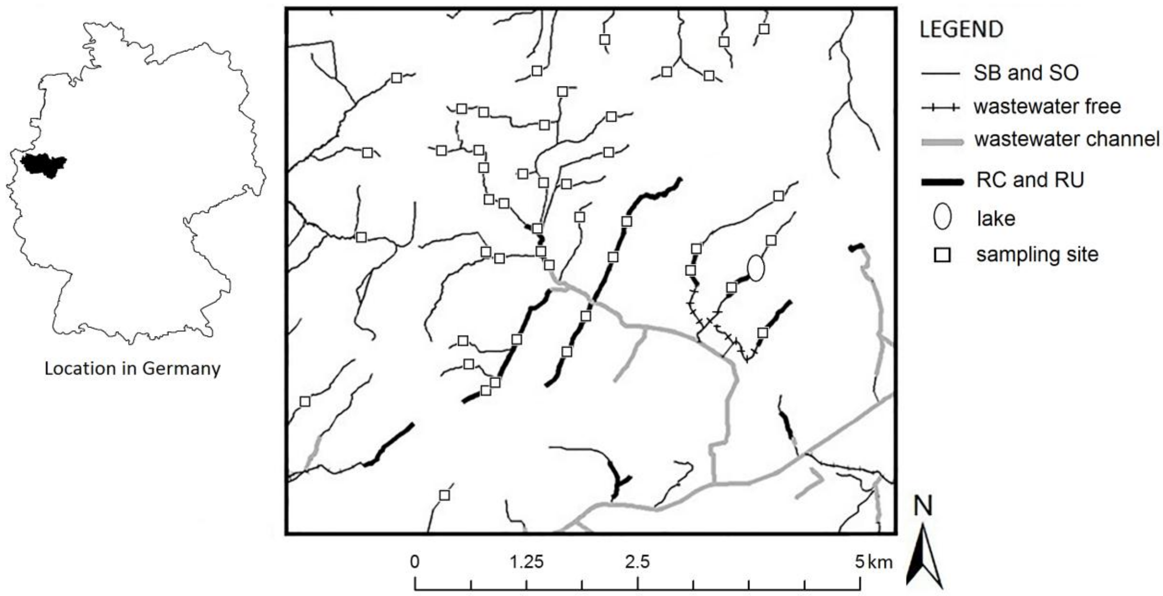

Figure 4.1 The study area of the Boye catchment including the locations of the sampling sites and its location in Germany. SB = source stream section within Boye catchment; SO = source stream section outside Boye catchment, RC = restored stream section connected to upstream near-natural stream section; RU = restored stream section unconnected to any upstream near-natural stream section.

4.2.2 Sampling, sorting and identification of taxa

Benthic invertebrate samplings were performed according to the German standard protocol, which ensures low variability in sampling, sorting and the resulting taxa list (Haase et al. 2004). The 45 sites were sampled between mid-March and mid-April 2012 at low to medium flow conditions (compare chapter 3). The 13 restored sites were exclusively sampled again at the end of March 2013. At each site, samples were taken using the multi-habitat-sampling technique (Haase et al. 2006) using a hand net of 25 x 25 cm (500 µ mesh size). The samples were sorted in the field and subsequently identified to the lowest feasible level in the laboratory (according to Haase et al. 2011). The resulting taxa lists were adjusted prior to analysis (Nijboer & Schmidt-Kloiber 2004).

4.2.3 Comparison of sites

Non-metric multidimensional scaling (NMS) was applied to the assemblages of all sites and (in case of the restored sites) of both years. First, the NMS was based on Jaccard similarity index reflecting the presence/absence of taxa as a proxy for successful dispersal; second, it was based on Bray-Curtis similarity (using log+1 transformed data) as a proxy for population establishment. One outlier source site was excluded from both NMS analyses. Differences between the assemblages of the water body groups (RC, RU, SB, SO) and between young restored sites (1-5 years) and old restored sites (9-20 years), as well as between young, connected and young, unconnected sites, were tested with an analysis of similarity (ANOSIM). The similarity of the assemblages of the four waterbody groups, of the young and old restored sites, and young, connected and young, unconnected sites were further compared using the Bray-Curtis index (using log+1 transformed data) (SIMPER; similarity percentages).

4.2.4 Changes in taxonomic composition after 1 year

The change of the taxonomic composition after 1 year was used as a proxy for the maturation of restored sites. We calculated a "change-value" from the NMS results. The algorithm of a NMS arranges each assemblage in a *N* dimensional space to preserve the distances between assemblages as much as possible, resulting in a vector distance matrix, which reflects the similarity of assemblages.

We used the coordinates of each site in the 3D vector NMS matrix to calculate two different "change values" for the Jaccard and Bray-Curtis based similarities:

(1) the distance (d) between the coordinates of the assemblage of a restored site sampled in 2012 (a1-a3) and in 2013 (b1-b3) in the 3D vector NMS matrix:

$$d = \sqrt{(b1-a1)^2+(b2-a2)^2+(b3-a3)^2}$$

(2) the distance (d) between the coordinates of the assemblage of a restored site (years 2012 and 2013 separately) (a1-a3) and the coordinates of the centre of the source colonisation assemblages (SB and SO), whereas the centre of source assemblages (c1-c3) is the mean of the coordinates of all source assemblages in the 3D vector NMS matrix:

$$d = \sqrt{(c1-a2)^2+(c2-a2)^2+(c3-a3)^2}$$

Due to the lower stress (0.17) in the 3D ordination as compared to the 2D matrix (0.23), we calculated the distances in the 3D matrix, but displayed the 2D ordination in the result chapter. A low change value (range: 0-2) indicates small changes in taxonomic composition and thus, a stable assemblage or low recolonisation from the surrounding. The change values were compared between the restored sites as a function of time since restoration (linear regressions).

4.2.5 Influence of local conditions on the recolonisation success

Environmental conditions at the restored sites were correlated (Spearman´s rho) with the following response variables: number of new taxa, number of lost taxa and number of steady taxa at the sites (comparing 2012 and 2013), Jaccard similarity and Bray-Curtis similarity between 2012 and 2013. According to our hypotheses 24 explanatory environmental parameters were addressed (Table 4.5):

- Time since restoration: number of years between the finalisation of the restoration measures and the sampling;
- Length of streams in a circle of 1 km and 5 km radius around the restored sites (according to Winking et al. 2014; Sundermann et al. 2011a); only source streams or older restored streams were considered.
- Share of five different land uses in a circle around the sites of 1 km and 5 km radius [%]: These parameters were calculated in GIS/ArcMap 10 (ESRI, Redlands, USA) on the basis of ATKIS® Data. The 46 different types of land use according to ATKIS® were categorised and merged into the five land use classes: urban use, agricultural area, grassland, forest (coniferous and deciduous forest excluding riparian vegetation) and area of surface waters (including streams, ditches, ponds and lakes). Besides, the land use was regarded as a proxy for water chemistry (Blann et al. 2009).
- Share of five different land uses in a buffer of 40 x 600 m [%]: According to Kail & Hering (2009) the morphology of 500 m and 1,000 m upstream sections significantly influence benthic invertebrate assemblages; for our small study sites we used 500 m upstream length and 20 m buffer width on each stream site. Buffers were generated in GIS/ArcMap 10 (ESRI, Redlands, USA). Additional 100 m downstream length and 20 m width on each site were added to the buffer to consider movement of invertebrates from downstream. The share of land use was calculated as described above.
- Share of seven microhabitats [%], recorded as a basis for multi-habitat-sampling: sand/sludge, gravel/stones, loam, living parts of terrestrial plants (LPTP), macrophytes (submerged and emergent), particulate organic matter (POM, coarse and fine) and algae. As the changes in the share of microhabitats from 2012 to 2013 were minor we used the share of microhabitats of the youngest sampling event (2013).

Altogether 120 different combinations of the 24 explanatory environmental parameters and the five response variables were correlated in Statistica 12.5 (Tulsa, Oklahoma, USA). The analysis normally required a p-value correction to minimise the type I error appropriate to multiple comparisons. However, the conservative Bonferroni correction increases type II errors. Thus, significant levels were provided uncorrected, because no correlation remained significant at $p > 0.05$ after the p-value correction by Bonferroni or the less conservative FDR (false discovery rate by Benjamini & Hochberg 1995). Instead, the scatterplots of the correlation analysis were visually checked and we further checked gradient lengths.

4.3 Results

4.3.1 Overall assemblage information

In the year 2012 we recorded 77 Taxa, in 2013 we recorded 62 Taxa at the restored sites. On average 20 Taxa were found at restored connected sites (RC, n=7; range: 17-31) in the year 2012, whereas in 2013 we found 19 Taxa (range: 9-13) (mean abundance 2012: 1,263 ind. m^{-2}; range: 610-4,128 and 2013: 850 ind. m^{-2}; range: 221-1,791). At the restored unconnected sites (RU, n=6), on average 17 taxa were recorded in 2012 (range: 11-36) with a mean abundance of 962 ind. m^{-2} (range: 580-1,443) and in 2013 14 taxa were found on average (range: 9-14) with a mean abundance of 847 ind. m^{-2} (range: 478-1,230 ind. m^{-2}).

At the 32 source sites, which were sampled in 2012, 113 taxa were found (compare chapter 3). At source sites within the Boye catchment (SB) (n = 21), 21 taxa were recorded on average (range: 14-29) with a mean abundance of 1,582 ind. m^{-2} (range: 344-8,915 ind. m^{-2}). At the source sites outside of the Boye catchment (SO) (n = 11), 18 taxa were found on average (range: 9-24) with a mean abundance of 765 ind. m^{-2} (range: 193-1,874 ind. m^{-2}).

The assemblage similarities within restored sites slightly increased from 2012 to 2013 (Table 4.1). The differences between restored sites and source sites are apparent in both NMS plots (Figure 4.2). Whereas the assemblages of old restored sites (9-20 years since restoration) were more similar to the source sites, the assemblages of the young restored sites (1-5 years since restoration) were well separated from the assemblages of the source sites. Besides, the restored unconnected sites are more strongly separated from the source sites as the restored connected sites.

Table 4.1 The qualitative (Jaccard) and quantitative (Bray-Curtis) average similarities of assemblage within source sites, and within restored sites and between source and restored sites compared between the years 2012 and 2013. For abbreviations of water body groups compare Figure 4.1.

	Jaccard [%]			Bray-Curtis [%]		
	SB+SO 2012	RC+RU 2012	RC+RU 2013	SB+SO 2012	RC+RU 2012	RC+RU 2013
SB+SO 2012	24.6	19.7	21.0	31.9	27.8	29.0
RC+RU 2012	19.7	25.3	25.7	27.8	36.1	36.8
RC+RU 2013	21.0	25.7	28.5	29.0	36.8	38.4

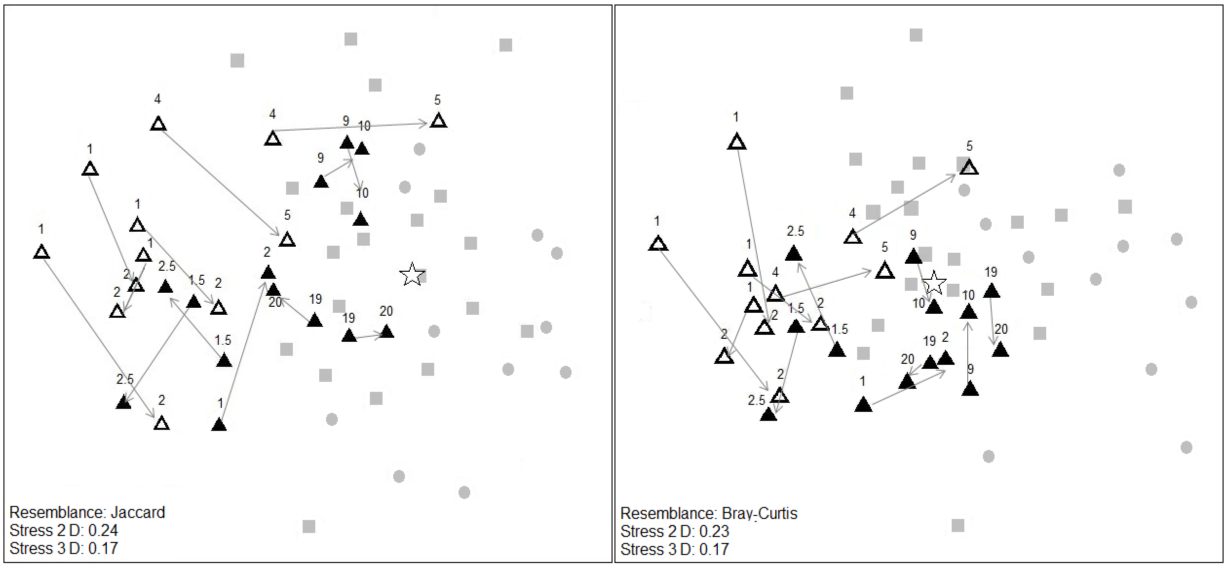

Figure 4.2 Non-metric multidimensional scaling (NMS) ordination of benthic invertebrate assemblages in sampling sites of the four water body groups (n = 57) based on Jaccard similarity and Bray-Curtis similarity: Full triangles = restored, connected site (RC); open triangles = restored, unconnected site (RU); grey rectangle = source within Boye catchment (SB); grey circle = source outside Boye catchment (SO). Numbers = years since restoration; arrows = change/distance of the same sampling site after 1 year; star = center of the source sites. Restored sites were sampled in 2012 and 2013, source sites only in 2012.

4.3.2 Changes in taxonomic composition after 1 year

The taxonomic composition of the old, connected restored sites sampled in 2012 resembled the composition of the source sites. Assemblage similarity of these sites to the source sites changed only slightly from 2012 to 2013 (Figure 4.2) and the number of steady taxa in these sites (mean: 14) is higher compared to the young, connected restored sites (mean 3.5) (Table 4.2). In general, assemblages of connected restored sites (RC) did not differ significantly from the source sites (SB and SO) in both years (ANOSIM based on both the Jaccard index and Bray-Curtis index; $R > 0.5$, $p \leq 0.05$, Table 4.3). In 2012, assemblages of unconnected restored sites (RU) were significantly different from the source sites within the Boye catchment (SB); this difference vanished from 2012 to 2013 (ANOSIM based on Jaccard index, Table 4.3). Besides, the unconnected restored sites (RU) differed significantly from the source sites outside the Boye catchment (SO) in both years (ANOSIM based on both the Jaccard index and Bray-Curtis index, Table 4.3), but the R-value decreased from 0.805 (Jaccard index)/0.752 (Bray-Curtis index) to 0.565 (Jaccard index)/0.587 (Bray-Curtis index) within 1 year.

The similarity between unconnected restored sites (RU) and the source sites (SB and SO) was higher in 2013 than in 2012, but still lower than the similarity between connected restored sites (RC) and the source sites (SB and SO) (SIMPER, Table 4.4). The similarity between the restored connected sites (RC) and the restored unconnected sites (RU) increased after 1 year, and the similarity of young, connected and young, unconnected sites showed the highest similarity values (2012: 39 %; 2013: 44 % (SIMPER, Table 4.4). The assemblages of young compared to the old restored sites differed in 2012 significantly from each other, but did not in 2013 (ANOSIM based on both the Jaccard index and Bray-Curtis index, Table 4.3).

Overall, the assemblages of the old restored sites changed less than those of the young restored sites after 1 year, whereas the variance of the young restored sites is due to one unconnected site with a small change value (linear regression of the “change value” - distance to the same site after 1 year, based on both the Jaccard index and Bray-Curtis index, Figure 4.3). The quantitative similarity of the young restored sites (1-5 years) to the source sites was higher after 1 year, whereas the quantitative similarity of old restored sites (9-20 years) to the source sites had barely changed (linear regression of the “change value” - distance to the centre of source sites 2012 vs. 2013, based on the Jaccard index, Figure 4.3). The qualitative similarity of the young restored sites (1-5 years) to the source sites was also higher after 1 year, but this is also true for old restored sites (9-10 years). Only the oldest restored sites (19-20 years since restoration) barely changed (linear regression of the “change value” - distance to the centre of source sites 2012 vs. 2013, based on the Bray-Curtis index, Figure 4.3).

4.3.3 Influences of environmental parameters on recolonisation success

Only few environmental parameters explained the variability of the biota´s change after 1 year. Significant correlations at $p < 0.05$ were found for: years since restoration, length of stream water surface in a radius of 1 km, share of water surface (including streams, lakes, ditches and ponds) in a radius of 5 km, share of urban use and forest in a radius of 1 km, share of urban use in a buffer of 600 x 20m, share of the microhabitats gravel/stones and macrophytes in the stream bed (Table 4.5).

Table 4.2 Taxonomic chance of the restored sites after 1 year, including the number (#) of new taxa, lost taxa, steady taxa and the Jaccard and Bray-Curtis similarities of each site between 2012 and 2013. RC = restored, connected site. RU = restored, unconnected site.

Waterbody group of site	Age of restoration in 2013	# New taxa after 1 year	# Lost taxa after 1 year	# Steady taxa after 1 year	Jaccard similarity [%] (2012 to 2013)	Bray-Curtis similarity [%] (2012 to 2013)
RC	20	13	3	14	48.49	46.93
RC	20	4	4	11	50	63.56
RC	10	10	5	11	42.86	60.89
RC	10	8	8	20	56.41	64.41
RC	2.5	6	12	4	21.74	39.72
RC	2.5	3	16	4	23.08	41.35
RC	2	5	10	6	30.44	41.13
RU	5	10	11	6	20.69	39.359
RU	5	7	16	8	27.27	42.54
RU	2	6	9	8	37.5	47.03
RU	2	5	11	3	20	38.75
RU	2	7	5	7	33.33	40.59
RU	2	4	3	7	53.33	67.97

Table 4.3 Pairwise assemblage comparison of the water body groups and of old and young restored sites (ANOSIM based on Jaccard index and on Bray-Curtis index) based on data from 2012 and from 2013. If R-value equals 1: similarity within the groups is higher than similarity to sites of the other group (R > 0.75: well separated; R > 0.5: overlapping, but clearly different; R ≤ 0.5: barely separable). For abbreviations of water body groups compare Figure 4.1.

	Jaccard		Bray-Curtis	
Groups	2012 R-value	2013 R-value	2012 R-value	2013 R-value
(1) RC, (2) RU	0.381*	0.043	0.397*	-0.04
(1) RC, (3) SB	0.174*	0.173	0.119	0.151
(1) RC, (4) SO	0.323*	0.325*	0.36*	0.359*
(2) RU, (3) SB	0.527*	0.334*	0.477*	0.295*
(2) RU, (4) SO	0.805*	0.565*	0.752*	0.587*
(3) SB, (4) SO	0.128*	-	0.269*	-
(5) young, (6) old	0.618*	0.455*	0.569*	0.417*
(7) young-RC, (8) young-RU	0.130	0.093	0.167	0.111

Table 4.4 Pairwise Bray-Curtis average similarity of the water body groups and of old and young restored sites (SIMPER). For abbreviations of water body groups compare Figure 4.1.

	2012	2013
Groups	Average similarity [%]	Average similarity [%]
(1) RC, (2) RU	32.20	38.20
(1) RC, (3) SB	34.67	33.32
(1) RC, (4) SO	28.05	26.38
(2) RU, (3) SB	26.81	30.52
(2) RU, (4) SO	16.72	20.31
(3) SB, (4) SO	29.85	-
(5) young, (6) old	30.00	34.62
(7) young-RC, (8) young-RU	39.18	43.82

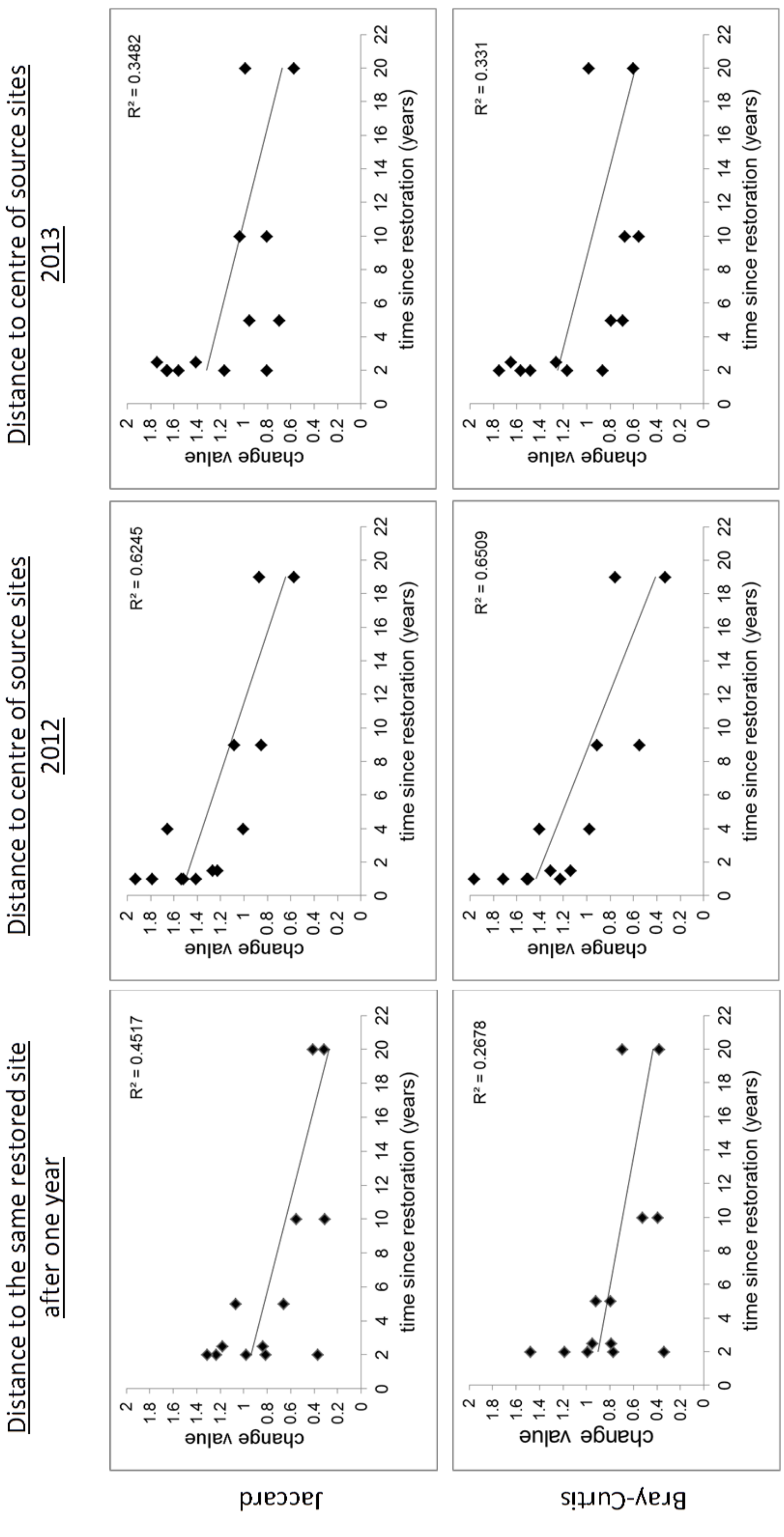

Figure 4.3 Linear regression of the "change value" for the restored sampling sites (n=13) calculated with the results of the 3D NMS ordination.

Table 4.5 Significant correlations (Spearman´s rho at $p < 0.05$) of land use parameters in a radius of 5 km and a buffer area of 40 x 600m to new taxa, lost taxa, steady taxa and similarity changes (Jaccard, Bray-Curtis) in restored sites in comparison of 2012 and 2013.

		New taxa	Lost taxa	Steady taxa	Jaccard	Bray-Curtis
[y]	Years since restoration			0.708*		0.610*
[km]	Stream water surface 5 km					
[km]	Stream water surface 1 km			0.582*	0.708*	0.726**
LU in 5 km radius [%]	Water surface	0.602*	-0.629*			
	Urban use					
	Agricultural area					
	Grassland					
	Forest					
LU in 1 km radius [%]	Water surface					
	Urban use					-0.615*
	Agricultural area					
	Grassland					
	Forest				0.650*	0.650*
LU in buffer [%]	Urban use				-0.580*	-0.636*
	Agricultural area					
	Grassland					
	Forest					
Share of micro habitats [%]	Gravel/Stone		-0.583*			
	Sludge/Sand					
	Loam					
	LPTP					
	Macrophytes		0.860**	-0.783**	-0.824**	-0.713**
	Algae					
	POM					

4.4 Discussion

4.4.1 Succession and maturation

As suggested by hypothesis H4, we observed further succession of the restored sites within 1 year, with larger changes of the assemblages of the young restored sites (1-5 years) than the assemblages of old (9-20 years) restored sites (NMS, “change value”). The assemblages of the young restored sites have not yet maturated to a degree that they resemble the source sites or the old restored sites. The first colonisers of the restored sites are pioneer assemblages consisting of well dispersing eurytopic taxa (chapter 3; Lorenz et al. 2009); the resulting assemblages are instable and characterised by a high change rate. Thus, individual species colonised the restored sites, but did not establish persisting populations. With progressing succession, the assemblages of sites restored 4 or 5 years ago are increasingly similar to the source sites. These results imply that a taxonomic maturation starts about 5 years after restoration. Similar time spans were found by Narf (1985), who showed that the aquatic invertebrate fauna in a relocated stream reach, which remained connected to upstream and downstream recolonisation sources, developed a fauna similar to control sites after 5.5 years. The direct connection to sources sites is vital for recolonisation and maturation, as shown by different initial recolonisation patterns in the first 3 years in connected sites compared to unconnected sites. While generalistic taxa with a high dispersal capability in the adult winged stage prevail in all freshly restored sites, they are supplemented by some hololimnic species in sites connected to upstream recolonisation sources, which colonise the restored sites mainly by drifting (chapter 3). This pattern was also hypothesised by Parkyn & Smith (2011), who stated that recolonisation of restored sites without upstream connections, is more slowly than in connected sites. Depending on stream size, Parkyn & Smith (2011) suggest 10-50 years for the maturation of connected restored sites. This is in line with our results, as only the assemblages of the very old restored sites (20 years since restoration) are similar to the source sites.

We expected the succession to equalise assemblages of connected and unconnected restored sites concerning their similarity to source sites. This was not clearly supported by our results. In 2012, the assemblages of the restored unconnected sites (RU) were significantly different from the source sites. Contrastingly, in 2013, they did not differ from the source sites within the Boye catchment (SB), but just from the source sites outside the Boye catchment (SO). Responsible for this change were mainly the old (4 or 5 years) unconnected restored sites, which assimilated to those of the source sites in 2013, though there are still differences. In contrast to the connected sites, the supply of colonists to unconnected sites takes more time and is mainly related to strong dispersers or passive random events such as floods, wind, attachment to animals, dispersal through groundwater or unintentional introductions by humans (Parkyn & Smith 2011; Briers et al. 2003; Van Leeuwen et al. 2013; Boulton et al. 1998). Besides, there also might be tendencies that the connection is less relevant for the recolonisation after a certain period of time or when the streams environmental conditions (like catchment land use, riparian habitats or in-stream habitats) are appropriate for sensitive taxa. In summary, a direct connection to source sites might be more important for rapid primary colonisation with hololimnic species and less mobile species.

4.4.2 Impact of local conditions on recolonisation success

We expected environmental parameters to steer the recolonisation process and the state of maturation (hypothesis H5). Generally, this hypothesis is supported by our results. The number of stable taxa is significantly higher (Mann-Whitney U-test) in connected restored sites than in unconnected restored sites (Table 4.2), which indicates a support of the succession by recolonisation sources in the surrounding; thus, the assemblages of unconnected sites are more variable and colonised by less additional taxa after 1 year. Furthermore, the number of new species at a restored site compared to the year before is positively correlated with the area of water surfaces in a radius of 5 km, including streams, lakes, ponds and ditches (Table 4.5). However, there was no correlation between the number of new species and the length of stream water surfaces, indicating that both standing waters and young restored streams are typical habitats of r-strategists, i.e. species coping with instable conditions. Furthermore, both standing waters and young restored streams have similar habitat conditions such as lack of riparian vegetation leading to warm water and macrophytes growth. Both factors are favoured by certain species like e.g. *Limnephilus lunatus* or *Mystacides azurea*, which were found in the restored sites and have a high dispersal capability. Early successional stages were also found by Lorenz et al. (2009) in warm, open and slowly flowing streams.

The number of lost species is negatively correlated with the water surface in a radius of 5 km. Therefore, recolonisation sources in the surrounding of a restored site stabilises the assemblage by enhancing the regional species pool (e.g. Hughes 2007; Lake et al. 2007; Jähnig et al. 2010). In isolated restored sites species` dispersal capability is a crucial factor for recolonisation (Cushing & Gains 1989; Milner et al. 2000). Tonkin et al. (2014) showed that it is not only the presence of taxa in the surrounding but also their prevalence affecting the probability of recolonisation.

We expected unstable assemblages at restored sites characterised by a high share of urban land use in the surrounding. Our results support this hypothesis: Bray-Curtis similarity between 2012 and 2013 was low in case of high urban land use (in a radius of 1 km and in a buffer of 40 x 600 m (Table 4.5). The Jaccard similarity of 2012 and 2013 assemblages were correlated to the share of urban use in the buffer of 40 x 600 m but not to urban use in a radius of 1 km. This can be a hint that urban use in the surrounding of 1 km limit the establishment of taxa (proxy Bray-Curtis similarity), but not their dispersal (proxy Jaccard similarity). The share of urban land use in the radius of 5 km, however, seems to be of minor importance as there was no correlation to both similarity indices.

Similar to our results, Parkyn et al. (2003) showed that biotic indices are only slowly improving, even in restored sites which are up to 20 years old, if the catchment upstream is fragmented or agriculturally used. Our results are adding to the growing body of literature on the role of riparian land use for in-stream habitats and biota (e.g. Death & Collier 2010; Jones et al. 2001; Kiffney et al. 2003). Especially urban land use is associated with several stressors (e.g. diffuse inputs, morphological degradation, hydraulic stress) (Paul & Meyer 2001, Walsh et al. 2005) leading to unstable, less predictable and less favourable conditions (Wahl et al. 2013). Finally, we expected microhabitat conditions of restored sites to impact the stability of assemblages with sludge and sand being unfavourable. This hypothesis was mainly rejected, as we found no significant correlation between the share of sludge/sand and assemblage development. This outcome is maybe due to a minor gradient as sand is the most dominant habitat in all sites. However, the share of gravel and stones was negatively correlated to the number of lost species. Besides dead wood, gravel and stones are the only stable microhabitats in sand-bottom streams, acting as refugia from hydrological stress, as clinging habitats for sessile and semi-sessile filter feeders and as feeding grounds for grazers utilising biofilms (Beisel et al. 2000). Thus, the establishment of species in restored streams is associated to the presence of key habitats (Lorenz et al. 2009).

In our study, sites rich of macrophytes were characterised by less stable assemblages: more taxa got lost, less were steady and the assemblages were less similar between 2012 and 2013. This is probably a result of the absence of riparian vegetation and shade, which might more directly influence the stability of the assemblage than macrophytes itself. Furthermore, the absence of riparian vegetation leads to higher water temperatures and to higher daily fluctuations of the oxygen content. In conclusion, not only the recolonisation and establishment of invertebrates needs time, but it is also needed for the development of in stream habitat conditions (Collier et al. 2001).

5 Summary, conclusions and future prospects

5.1 Background

This thesis aimed at the ecological evaluation of restored former sewage channels located in the highly urbanised Emscher catchment in western Germany. Prior to restoration the study streams had been used as open sewers for decades and benthic invertebrate life was not possible except for some sewage tolerant Oligochaeta. Restoration measures included the construction of underground sewers for the wastewater, the near-natural remodeling of riparian areas and of the stream bed.

The unique situation in the streams of the Emscher catchment allowed to investigate the recolonisation of restored urban streams by benthic invertebrates and follow-up, the restoration success and additionally, the primary factors influencing the recolonisation.

According to literature the following factors predominantly influence recolonisation: the recolonisation potential (e.g. Sundermann et al. 2011a; Tonkin et al. 2014), the species dispersal capability (Cañedo-Argüelles et al. 2015), the environmental conditions and landscape context (Hughes et al. 2008; Reynolds et al. 2013) and the succession processes (McCook 1994). These influencing factors were mainly investigated in streams of the open landscape. Therefore, their influence on the urban streams of the Emscher catchment was analysed in this thesis. For this purpose, new indicators were developed and indicators of the Water Framework Directive (WFD) were used. The WFD aims at improving the chemical and biological quality of European waters. For heavily modified water bodies, like the streams of the Emscher system, the goal is to reach the “Good Ecological Potential” until 2015 (or under certain circumstances until 2021 or 2027) (European Commission 2000).

In summary, this thesis focussed on the ecological evaluation of the three main topics: First, the ecological assessment of the restored streams after restoration according to the WFD and environmental parameters influencing the Good Ecological Potential; second, the primary colonisation by benthic invertebrates after restoration and their recolonisation patterns; and third, the succession of benthic invertebrates communities and environmental parameters steering the succession process.

5.2 Ecological assessment (chapter 2)

The thesis starts with an overview of the ecological status of restored sites in the whole Emscher catchment. Based on 248 taxa lists of benthic invertebrates sampled in restored sites by different sampling methods, the analysis focused on the Ecological Potential according to the requirements of the WFD. As possible explanatory parameters for the Ecological Potential, amongst others, riparian land use, stream habitats, and time since restoration were included into a PCA analysis.

Almost 40 % of the sites already achieve the Good Ecological Potential at the recent sampling. Environmental parameters enhancing the probability of meeting the Good Ecological Potential include: connection of restored sites to an unmodified stream section upstream, dead wood in the stream bed, good hydro-morphological structure, deciduous riparian vegetation and unsealed surface in the stream´s surrounding, while the occurrence of iron ochre and sewage overflows located upstream of the sampling sites hinder the achievement of the Good Ecological Potential.

This study also reveals that the Ecological Potential of the restored streams in the Emscher system is only indirectly determined by the factor time. Although the statistical analysis presented this factor as most influencing, in the minority of cases a parallel development of the Ecological Potential and time was found. Instead, the above mentioned environmental conditions of a restored site are of greater importance for the assessment and the achievement of the Good Ecological Potential. In conclusion, the factor “time since restoration” must be interpreted as a proxy for the overall development of the (aquatic and terrestrial) habitats in and at the restored sites.

5.1 Recolonisation (chapter 3)

The recolonisation success and a good ecological assessment of the streams are amongst others dependent on recolonisation sources and the dispersal capabilities of taxa. Therefore, in a second step I analysed the recolonisation processes by benthic invertebrates. For this analysis, the case study catchment of the Boye, a 77 km² sub-catchment of the Emscher was chosen. This catchment has a high number of restored streams with almost the same ecological conditions as the whole Emscher catchment.

In the spring 2012, seven restored sites connected to near-natural upstream sections were sampled, which were never used as sewage channels and are in good status morphologically. Furthermore, six unconnected restored sites were sampled. Restoration measures had been conducted between one and 19 years before sampling. Additionally, 21 near-natural sites within the catchment and eleven near-natural sites in neighbouring catchments were sampled. Near-natural sites were considered to be potential source sites from which benthic invertebrates might colonise the restored sites. 128 taxa were recorded and were categorised into five dispersal classes reflecting dispersal capabilities and degree of ecological specialisation, according to a literature review. Assemblages at restored sites were characterised by lower numbers of taxa and/or high abundances of hololimnic taxa and poorly dispersing winged species and by higher species numbers and abundance of strongly dispersing generalists. A recolonisation sequence was derived from the observed patterns, in which winged, strongly dispersing generalists colonised most rapidly and were followed by hololimnic species, weakly dispersing generalists and habitat specialists. Restored sites connected to near-natural upstream sections were colonised more rapidly than unconnected restored sites, particularly by habitat specialists.

Almost 90 % of the recolonisation events originated from sources within a distance of 5 km. A succession from pioneer assemblages to more mature communities, which resembles that of the surrounding near-natural sites, was observed. In summary, assemblages in connected, restored sites needed 9 to 19 years to reach maturation, while the settlement of assemblages in unconnected sites are expected to require more time.

5.2 Succession (chapter 4)

While successional changes of assemblages in lakes or wetlands are well documented, these processes are poorly understood in streams. Following stream restoration and primary recolonisation the benthic invertebrate assemblage is also supposed to undergo a succession, as new habitats have been generated. These successional changes are important to predict the taxonomical development, thus indirectly the development of the ecological assessment, of restored sites. Therefore, the same 13 sites in the seven restored streams as of the Boye sub-catchment in chapter 3 were investigated again in the spring 2013. For each site environmental parameters expected to steer the succession process were collected. Their influence on the inter-annual taxonomical change was tested with correlation analyses (Spearman’s rho). The 21 near-natural sites within the Boye catchment and 11 near-natural sites in neighbouring catchments sampled in 2012 served again as source sites for the analysis.

Within 1 year time, the restored sites have undergone further succession, which lead to a higher resemblance of their assemblages to those of the source sites. These results were derived from similarity analyses, non-metric multidimensional scaling and therefrom developed change values, which show the taxonomical change of a site after 1 year dependent of the recolonisation sources. The assemblages of young restored sites changed more markedly than assemblages of old restored sites within the time span of 1 year. In the first years after restoration instable assemblages with high abundances of eurytopic pioneer taxa were found, while 5 years after restoration assemblages were increasingly similar to those of the source sites and mature assemblages were observed 9 to 10 years after restoration. Differences between young restored sites connected and unconnected to near-natural upstream sections were observed, suggesting a strong colonisation with organisms from upstream sections particularly in the first years.

The succession towards near-natural assemblages is further supported by recolonisation sources in the surroundings, the presence of gravel/stones at the stream bottom and a low share of urban land use in the surrounding. Especially urban land use is associated with several stressors (e.g. diffuse inputs, morphological degradation, hydraulic stress) leading to unstable, less predictable and less favourable conditions.

5.3 Conclusion and future prospects

From the results of the first study (chapter 2), suggestions, as the creation/enhancement of growth of deciduous woody riparian vegetation along buffer strips of the streams, the reduction and improvement of sewage overflows, the provisioning of a connection to the streams tributaries, and active addition of dead wood to the streams, for further optimisation of the restoration of urban streams were derived. They serve as recommendations to improve restoration measures in the future. Furthermore, the second study (chapter 3) showed that the establishment of mature habitat conditions, in particular woody riparian vegetation, is a prerequisite for the recolonisation of habitat specialists, which indicate the maturation progress of a restored site. In the planning phase of a restoration this knowledge can be used to especially create habitats in order to promote the recolonisation of sensitive species. Several streams of the Emscher catchment are still isolated after restoration and not directly connected to colonisation sources. For these streams a possible approach might be an assisted migration of invertebrates, which would not reach these streams on their own. This can probably help to reach the target assemblage and the Good Ecological Potential in due course.

The results of the second and third study (chapters 3 and 4) suggest that the invertebrate assemblages will reflect the restoration effects in sense of maturation at earliest 5 years, but more likely a decade, after restoration. Applying the “change value” used in the third study, timeframes for the monitoring of restored sites can be aligned for each river network, dependent of the recolonisation sources of the surrounding and the environmental conditions of the study streams´ catchment. My findings about succession help to set realistic goals in stream restoration. It can be detected at an early stage, whether the restoration target or the achievement of the Good Ecological Potential is realistic for a stream or not.

Often, the success of a restoration measure initially does not result in the Good Ecological Potential. Nevertheless, it is important to know, whether a site has reached a “small success”, in terms of a taxonomic improvement, after a restoration measure. For this purpose, the dispersal classes of the second study can be used. The dispersal classification is transferable to other catchments and additional taxa could be classified following the described rules. As the number of taxa representing different dispersal classes is more constant than species richness or similarity patterns, dispersal classes could be used as a generic trait to analyse assemblage maturation. Small successes like the recolonisation of demanding taxa can get apparent. To better understand the processes of recolonisation and succession, long term studies, ideally also addressing population genetics, are advisable in the future.

6 Zusammenfassung

Ökologische Auswertungen renaturierter, ehemaliger Abwasserkanäle im urban geprägten Emscher-Einzugsgebiet

Hintergrund

Urbane Fließgewässer unterscheiden sich in vielerlei Hinsicht von Bächen und Flüssen der freien Landschaft: ihre laterale Entwicklung ist aufgrund von Verkehrswegen, Leitungen und Bebauung eingeschränkt, die Hydromorphologie und die Wasserqualität sind durch vielfältige Belastungen beeinträchtigt. Zudem sind häufig lange Strecken verrohrt, so dass die verbleibenden Abschnitte isoliert sind (Bernardt & Palmer 2007). Diese Faktoren wirken sich stark auf die aquatischen Lebensgemeinschaften (Zönosen) in urbanen Gewässern aus und führen häufig dazu, dass diese durch einen hohen Anteil an anspruchslosen Arten und einen geringen Anteil störungssensibler Arten gekennzeichnet sind (Coles et al. 2012). Renaturierungsmaßnahmen zur Verbesserung der biologischen Situation sind häufig kompliziert und kostspielig. Dennoch ist die Zahl der Renaturierungsprojekte zur ökologischen Verbesserung der Gewässer in urbanen Regionen seit den letzten 20 Jahren deutlich angestiegen (Walsh et al. 2005). Das Wissen über den Effekt von Renaturierungsmaßnahmen in urbanen Gewässern auf die Gewässerökologie wird zwar stetig erweitert, ist jedoch noch immer begrenzt (Kenney et al. 2012). Dementsprechend ist es wichtig, den Einfluss einer verbesserten Wasserqualität und Morphologie auf den ökologischen Status in urbanen Gewässern detailliert zu untersuchen um zukünftige Renaturierungsmaßnahmen optimieren zu können.

Zur Untersuchung von renaturierten[1], urbanen Fließgewässern eignen sich besonders die Gewässer des Emscher-Einzugsgebiets im Ruhrgebiet. Es handelt sich um ein

[1] Der eigentlichen Wortbedeutung nach ist der Begriff „Renaturierung" für die urbanen Gewässer des Emschersystems nicht richtig und zu unpräzise, denn der natürliche Zustand kann auch nach Renaturierungsmaßnahmen nicht wieder hergestellt werden. Treffender wäre es daher von einer „ökologischen Verbesserung" oder einer „ökologischen Aufwertung" zu sprechen. Zur Vereinfachung und besseren Lesbarkeit des Textes wird aber dennoch das Wort „Renaturierung" als Synonym für „ökologische Verbesserung" verwendet.

Einzugsgebiet im Westen Deutschlands, das bei einer Gesamteinwohnerzahl von 2,4 Millionen auf 865 km² und ca. 2800 Einwohner pro km² eine hohe Besiedlungsdichte aufweist. Im Vergleich zu Gewässern anderer urbaner Räume sind die Gewässer des Ruhrgebietes besonders stark beeinträchtigt: in Betonhalbschalen verlegt, dienten sie stellenweise seit 100 Jahren dem oberirdischen Transport von Abwasser und waren, abgesehen von Würmern, nicht besiedelt. Aufgrund des Bergbaus und den damit einhergehenden Bergsenkungen war es lange Zeit nicht möglich unterirdische Abwasserkanäle zu verlegen. Große Teile der Emscher und Teile ihrer Nebengewässer wurden daher um 1900 auf einer Länge von 350 km zu oberirdischen Abwasserkanälen ausgebaut. Dies entspricht mehr als der Hälfte der Gesamtlänge der Fließgewässer des Emscher-Netzwerks (ca. 640 km). Die verbleibenden 290 km blieben vom Kanalbau unbetroffen und waren durchgehend mit aquatischen Lebensgemeinschaften besiedelt.

Der Untergrund des Ruhrgebietes ist zudem ein Flickenteppich aus Altlasten, die Gewässer mit Schadstoffen aller Art belasten können. Straßenabwässer und eine hohe Anzahl an Regenüberläufen tragen ebenfalls zu einer komplexen Belastungssituation bei (EGLV 2015).

Mit dem Rückgang des Bergbaus zu Beginn der 1990er Jahre und dem damit einhergehenden Abklingen der Bergsenkungen, hat die Emschergenossenschaft damit begonnen, die Emscher und ihre Nebengewässer zu renaturieren: es werden Kanäle zum unterirdischen Transport des Abwassers gebaut, woraufhin die Betonhalbschalen, in denen die Gewässer verlaufen, entfernt und das Gewässerbett und das direkte Gewässerumfeld möglichst naturnah gestaltet werden. Insgesamt wurden bis heute ca. 123 Fließkilometer auf diese Weise renaturiert. Die weitere Planung des Emscher-Umbaus sieht vor, insgesamt 350 km zu renaturieren. Mit einem Investitionsvolumen von 4,5 Milliarden Euro handelt es sich um eines der größten Projekte zur naturnahen Umgestaltung eines Gewässersystems Europas (EGLV 2015).

Aufgrund ihrer besonderen Belastungen werden die Fließgewässer des Emschersystems, sowie die meisten urbanen Gewässer in der Gewässerbewertung und -bewirtschaftung im Sinne der europäischen Wasserrahmenrichtlinie (European Commission 2000) als „erheblich veränderte Gewässer“ ausgewiesen. Bei erheblich veränderten Gewässern stehen gewässerspezifische, anthropogene Nutzungen im Vordergrund. Der „Gute Ökologische Zustand“, das ambitionierte Ziel der Wasserrahmenrichtlinie ist dadurch im Regelfall nicht erreichbar. Stattdessen ist das „Gute Ökologische Potenzial“ als wasserwirtschaftliches Ziel für erheblich veränderte Gewässer definiert und muss bis zum Jahr 2015 erreicht werden (oder unter bestimmten Voraussetzungen bis 2012 oder 2027). Die Gewässer des Emscher-

Einzugsgebiets, müssen also das Gute Ökologische Potenzial erreichen, was mit weniger strengen, auf die anthropogene Nutzung ausgerichteten Bewertungskriterien verbunden ist (LAWA 2013).

In der Bewertung von Fließgewässern gemäß der Wasserrahmenrichtlinie werden u.a. benthische Invertebraten als Indikatoren verwendet. Dabei handelt es sich um Kleinstlebewesen, wie beispielsweise Insektenlarven, Muscheln, Schnecken oder Krebse, die auf dem Gewässergrund oder in der Gewässersohle leben. Aufgrund ihrer speziellen Anpassungen (wie z.B. an die Wasserchemie und die Habitate im Gewässer) (Malmquist 2002) eignet sich diese Tiergruppe besonders, um Aussagen über die chemischen, biologischen und strukturellen Verhältnisse im Gewässer zu treffen (Hering et al. 2004). In der vorliegenden Arbeit habe ich mich auf diese Tiergruppe beschränkt.

Die Besiedlung durch benthische Invertebraten setzt ein, sobald ein Gewässer abwasserfrei und naturnah umgestaltet ist. Für die Gewässer des Emscher-Einzugsgebiet bedeutet das, dass jeder renaturierte Abschnitt gänzlich neu besiedelt werden muss und Populationen sich etablieren und entwickeln müssen, denn die ehemaligen Abwasserkanäle waren abgesehen von Würmern nicht besiedelt. Diese besondere Situation bietet einerseits die Gelegenheit Untersuchungen zur Wiederbesiedlung beginnend bei „Null" zu beobachten. Zudem herrscht in den meisten Wiederbesiedlungsstudien eine Unsicherheit, ob bei einer Beprobung vor der Renaturierung nicht erfasste Taxa tatsächlich nicht vorhanden waren oder ob sie bei der Beprobung übersehen wurden. Dieser methodische Fehler kann in den Emschergewässern ausgeschlossen werden.

Die Wiederbesiedlung der Emschergewässer ist jedoch mit Hindernissen verbunden: die benthischen Invertebraten müssen nach der Renaturierung aus benachbarten Einzugsgebieten oder aus den Gewässerabschnitten, die nie vom Verbau betroffen waren - sogenannten potenziellen Wiederbesiedlungsquellen - einwandern. Zudem beeinflussen viele weitere Faktoren auf verschiedenen räumlichen Skalen die Wiederbesiedlung und somit auch den Erfolg einer Renaturierungsmaßnahme bzw. deren ökologische Bewertung. Einen starken Einfluss auf die Wiederbesiedlung und damit den Renaturierungserfolg haben laut Literatur das Wiederbesiedlungspotenzial der renaturierten Gewässer, d.h. das Vorhandensein von Wiederbesiedlungsquellen in der Umgebung (z.B. Sundermann et al. 2011b; Tonkin et al. 2014), die Ausbreitungsfähigkeit der Taxa (Cañedo-Argüelles et al. 2015), die Umweltbedingungen in und an den Gewässern, z.B. Mikrohabitate der Gewässersohle, die Landnutzung, Auenvegetation etc. (Hughes et al. 2008; Reynolds et al. 2013) und

Sukzessionsprozesse (McCook 1994). Der Einfluss dieser Faktoren wurde jedoch vorrangig für Gewässer in der freien Landschaft untersucht. Wohingegen, deren Einfluss in der vorliegenden Arbeit detailliert für die urbanen Gewässer des Emscher-Einzugsgebiets untersucht wurde. Der Erfolg bereits durchgeführter Renaturierungsmaßnahmen wurde aus verschiedenen Blickwinkeln betrachtet, um Handlungsempfehlungen zur zukünftigen Optimierung von Renaturierungsmaßnahmen abzuleiten. Für die Einschätzung des Renaturierungserfolgs wurden sowohl Indikatoren der Wasserrahmenrichtlinie, als auch neue, selbst entwickelte Indikatoren verwendet. Hierzu wurde das Ökologische Potenzial der renaturierten Gewässerabschnitte im Emschereinzugsgebiet bestimmt und die Einflüsse von Umweltbedingungen auf die Bewertung der Gewässer, sowie Wiederbesiedlungsprozesse und die Sukzession der benthischen Zönosen untersucht.

Die vorliegende Arbeit gliedert sich daher in die folgenden drei Themenbereiche, deren Hauptergebnisse nachfolgend kurz zusammengefasst werden:

1) Die ökologische Bewertung der renaturierten Gewässerabschnitte und Einflüsse von Umweltparametern auf das Ökologische Potenzial
2) Die Wiederbesiedlung der renaturierten Gewässerabschnitte und die Ausbreitungsfähigkeit der erfassen Taxa
3) Die Sukzession der benthischen Zönosen in den renaturierten Gewässerabschnitten in Abhängigkeit von bestimmten Umweltparametern

1 Die ökologische Bewertung der renaturierten Gewässerabschnitte und Einflüsse von Umweltparametern auf das Ökologische Potenzial (Kapitel 2)

Nach der Renaturierung entwickeln sich in den Gewässern der Emscherregion wieder benthische Zönosen, deren Entwicklung und ökologische Bewertung in dieser ersten Studie in Abhängigkeit von verschiedenen Umweltvariablen untersucht wurden. Die Datengrundlage bildeten 248 Taxalisten benthischer Invertebraten aus renaturierten Gewässerabschnitten, die nach standardisierten Probenahmemethoden (Multi-Habitat-Sampling (MHS) und DIN-Beprobung) genommen wurden. Anhand dessen wurde das Ökologische Potenzial für erheblich veränderte Gewässer nach den Vorgaben der Wasserrahmenrichtlinie berechnet. Außerdem wurden anthropogene Stressoren wie z.B. die Landnutzung und abiotische Parameter wie z.B. vergangene Zeit seit der Renaturierung, die das Erreichen des Ökologischen Potenzials entweder fördern oder hemmen können, mit PCA-Analysen evaluiert.

Fast 40 % der untersuchten Probestellen erreichten bei ihrer jüngsten Beprobung bereits das Gute Ökologische Potenzial. Diese Zielerreichung wurde durch eine Anbindung an nie verbaute Nebengewässer gefördert, aus denen anspruchsvolle Arten den renaturierten Gewässerabschnitt per Drift besiedeln können. Darüber hinaus trugen folgende Faktoren zur positiven Entwicklung einer naturnahen Zönose bei: Totholz im Gewässer, eine gute Gewässerstruktur sowie Laubgehölze und nicht versiegelte Flächen im Gewässerumfeld. Das Auftreten von Eisenocker und das Vorkommen von Entlastungsbauwerken oberhalb einer Probestelle können das Erreichen des Guten Ökologischen Potenzials hingegen negativ beeinflussen. Diese Studie hat zudem gezeigt, dass das Ökologische Potenzial der Emschergewässer nur indirekt von dem Faktor „vergangene Zeit seit der Renaturierung" beeinflusst wurde: obwohl die statistische Analyse diesen als den einflussreichsten Faktor identifiziert hatte, konnte in den wenigsten Fällen eine parallele Entwicklung von Zeit und Ökologischem Potenzial festgestellt werden. Stattdessen waren die oben beschriebenen Umweltparameter von weitaus größerer Bedeutung für die Bewertung der renaturierten Abschnitte. Demnach muss der Faktor Zeit als ein Proxy verstanden werden, der die Entwicklung der Gewässerhabitate und die der Aue- und Randstreifenbereiche widerspiegelt.
Aus den Ergebnissen ließen sich folgende Stellschrauben definieren, um zukünftige Renaturierungen weiter zu optimieren: Schaffung/Förderung von Bewuchs mit Laubgehölzen entlang der Gewässer, Kontrolle von Entlastungsbauwerken, Anbindungen von Nebengewässern und Einbringung von Totholz.

2 Die Wiederbesiedlung der renaturierten Gewässerabschnitte und die Ausbreitungsfähigkeit der erfassen Taxa (Kapitel 3)

Das Erreichen des Guten Ökologischen Potenzials ist nicht nur abhängig von den genannten Umweltparametern der vorangegangenen Studie, sondern auch von vorhandenen Wiederbesiedlungsquellen in der Umgebung der renaturierten Gewässerabschnitte und der Ausbreitungsfähigkeit der Taxa. Deshalb wurden in der nächsten Studie die Wiederbesiedlungsprozesse der benthischen Invertebraten analysiert. Hierzu wurden nicht-metrische multidimensionale Skalierungen angewandt, Ähnlichkeitsanalysen durchgeführt, Ausbreitungsdistanzen von Taxa analysiert und ein selbst entwickelter Wiederbesiedlungsquotient errechnet.
Für diese Studie wurde ein Teileinzugsgebiet der Emscher, das Boye-Einzugsgebiet, als Fallbeispielgebiet gewählt. Die Gründe der Wahl waren dessen hohe Anzahl an renaturierten

Gewässerabschnitten auf kleinem Raum (Teileinzugsgebietsgröße: 77 km²) mit annähernd gleichen Umweltbedingungen auf Teileinzugsgebietsebene.

Im Frühjahr 2012 wurden im Boye-Einzugsgebiet Beprobungen nach der MHS-Beprobung durchgeführt. Es wurden dabei sieben renaturierte Stellen mit Anbindung an einen naturnahen, nie vom Kanalbau betroffenen Oberlaufabschnitt beprobt. Diese Oberlaufabschnitte befinden sich jeweils in einem morphologisch guten Zustand. Zudem wurden sechs renaturierte Stellen ohne diese Anbindung beprobt. Die Fertigstellung der Renaturierung dieser 13 Stellen liegt zum Zeitpunkt der Probenahme zwischen einem und 19 Jahren zurück. Darüber hinaus wurden 21 naturnahe, nie verbaute Stellen innerhalb des Boye-Einzugsgebiets und elf naturnahe, nie verbaute Stellen in benachbarten Einzugsgebieten in einem Radius von 5 km beprobt. Die naturnahen Stellen wurden als potenzielle Wiederbesiedlungsquellen betrachtet.

Insgesamt wurden bei der Beprobung 128 Taxa erfasst und anhand einer umfangreichen Literaturstudie in fünf Ausbreitungsklassen eingeteilt, die die Ausbreitungsfähigkeit und den Grad der ökologischen Spezialisierung (z.B. Habitatansprüche) widerspiegeln.

Die Zönosen der renaturierten Stellen sind durch eine geringere Anzahl/Abundanz hololimnischer Taxa und Taxa mit geringer Ausbreitungsfähigkeit gekennzeichnet. Außerdem wurde eine höhere Anzahl/Abundanz an Taxa mit hoher Ausbreitungsfähigkeit erfasst. Anhand der Wiederbesiedlungsmuster wurde eine Reihenfolge der Besiedlung beobachtet: Taxa ohne besondere Habitatansprüche (Habitatgeneralisten), aber mit hoher Ausbreitungsfähigkeit, besiedelten die renaturierten Stellen am schnellsten, gefolgt von hololimnischen Taxa und später von Habitatgeneralisten mit geringer Ausbreitungsfähigkeit. Zuletzt folgten Habitatspezialisten, die an bestimmte Habitate angepasst sind. Renaturierte Stellen, die eine Anbindung an einen nie verbauten, naturnahen Abschnitt haben, wurden insgesamt schneller und insbesondere schneller von Habitatspezialisten wiederbesiedelt. Fast 90% der Wiederbesiedlungsereignisse erfolgten aus Wiederbesiedlungsquellen die sich innerhalb eines Umkreises von 5 km befinden. Zudem wurde eine Entwicklung von Pionierzönosen hin zu reiferen Zönosen[2] beobachtet. Zusammenfassend kann man aus den Ergebnissen schließen, dass Zönosen aus renaturierten Abschnitten mit einer Anbindung an naturnahe Abschnitte ca. neun bis 19 Jahre benötigen, um sich den Zönosen naturnaher Abschnitte anzunähern. Dahingegen wird erwartet, dass die Besiedlung und Etablierung von

[2] „reife Zönosen" werden in diesem Text Zönosen renaturierter Gewässerabschnitte bezeichnet, die den Zönosen der naturnahen Wiederbesiedlungsquellen taxonomisch ähneln und sich entsprechend dem Artenpool der Umgebung angeglichen haben.

vergleichbaren Zönosen insbesondere mit anspruchsvolleren Taxa an Stellen ohne diese Anbindung mehr Zeit benötigt.

3 Die Sukzession der benthischen Zönosen in den renaturierten Gewässerabschnitten in Abhängigkeit von bestimmten Umweltparametern (Kapitel 4)

Nachdem die renaturierten Gewässer durch benthische Invertebraten wiederbesiedelt worden sind, wird erwartet, dass deren Zönosen natürlicherweise eine Sukzession durchlaufen. Es ist wichtig, diese sukzessionellen Veränderungen zu erforschen, um die taxonomische Entwicklung in renaturierten Gewässerabschnitten und somit auch deren ökologische Bewertung abschätzen zu können. Daher wurden die in 2012 beprobten 13 Probestellen aus renaturierten Abschnitten des Boye-Einzugsgebiets im Jahr 2013 erneut mit dem MHS-Verfahren beprobt und untersucht. Zusätzlich wurden Umweltparameter, von denen erwartet wurde, dass sie einen Einfluss auf die Sukzessionsprozesse haben könnten, für die renaturierten Stellen erhoben. Der Einfluss dieser Umweltparameter auf die taxonomische Veränderung innerhalb eines Jahres, wurde mit Korrelationen (Spearman´s rho) analysiert. Außerdem wurden die Daten der 2012 beprobten Wiederbesiedlungsquellen für diese Studie erneut herangezogen.

Innerhalb eines Jahres war eine Sukzession der benthischen Zönosen erkennbar, die dazu geführt hat, dass sich die Zönosen der renaturierten Stellen denen der nahgelegenen Wiederbesiedlungsquellen mehr angeglichen haben als im Jahr davor. Anhand von Ähnlichkeitsanalysen, nicht-metrischen multidimensionalen Skalierungen und daraus errechneten Veränderungswerten (taxonomische Veränderung der renaturierten Probestellen nach einem Jahr in Abhängigkeit der Wiederbesiedlungsquellen), wurde festgestellt, dass junge renaturierte Stellen (1-5 Jahre) sich taxonomisch innerhalb eines Jahres mehr verändert haben als Stellen, deren Renaturierung schon länger zurück lag (mindestens 9 Jahre). In den ersten fünf Jahren nach der Renaturierung wurden instabile Zönosen mit hohen Abundanzen von Habitateneralisten erfasst. Fünf Jahre nach der Renaturierung hingegen wurde eine steigende Ähnlichkeit der Zönosen renaturierter Stellen ohne Anbindung und den Wiederbesiedlungsquellen nachgewiesen. Junge Stellen mit Anbindung an naturnahe Abschnitte waren den Wiederbesiedlungsquellen taxonomisch ähnlicher als junge Stellen ohne die Anbindung. Drifteffekte aus nie verbauten Oberlaufbereichen spielen demnach besonders in den ersten Jahren nach der Renaturierung eine wichtige Rolle für die Wiederbesiedlung und die anschließende Entwicklung der Zönose und werden mit der Zeit weniger relevant. Zönosen, die sich taxonomisch den umgebenen Wiederbesiedlungsquellen angeglichen haben, wurden zwischen neun und zehn Jahren nach der Renaturierung beobachtet. Die Ergebnisse dieser Studie zeigten zudem, dass eine Sukzession hin zu einer reiferen Zönose allgemein durch nah gelegene Wiederbesiedlungsquellen unterstützt wird. Einen weiteren positiven Einfluss für die Entwicklung einer reifen Zönose haben das Vorkommen von Kies und Steinen im Gewässersubstrat und ein geringer Anteil an urban genutzter Fläche im Gewässerumfeld. Besonders die urbane Nutzung im Gewässerumfeld steht in Verbindung mit einer Vielzahl an Stressoren, die auf das Gewässer wirken und zu unstabilen und wenig vorhersehbaren Zönosen führen.

Schlussfolgerung und Ausblick

Aus den Ergebnissen der ersten Studie (Kapitel 2) konnten Vorschläge zur Optimierung von Renaturierungsmaßnahmen der noch nicht umgestalteten Emschergewässer, aber auch generell für urbane Gewässer abgeleitet werden. Es ist empfehlenswert, diese Vorschläge für zukünftige Gewässerrenaturierungen in den Maßnahmenkatalog aufzunehmen. Wie in der ersten Studie wurde auch in der zweiten Studie (Kapitel 3) deutlich, dass Laubgehölze am Gewässerrand eine Voraussetzung für die Wiederansiedlung mit anspruchsvollen Arten (Habitatspezialisten) ist und eben diese den Reifeprozess von Zönosen anzeigen. In der Planungsphase einer Renaturierungsmaßnahme können mit diesem Wissen beispielsweise gezielt Habitate geschaffen werden, um eine Wiederbesiedlung mit sensiblen Arten aus der Umgebung zu fördern. Für Gewässer, die auch nach der Renaturierung isoliert sind oder in deren Umgebung geeignete Wiederbesiedlungsquellen fehlen, ist es ratsam, über eine aktive Wiederansiedlung von benthischen Invertebraten nachzudenken, um die Wahrscheinlichkeit, das Gute Ökologische Potenzial zu erreichen, zu erhöhen.

Die Ergebnisse der zweiten und dritten Studie (Kapitel 3 und 4) verdeutlichen, dass sich ein Renaturierungseffekt im Sinne einer Annäherung der Lebensgemeinschaften renaturierter Stellen an die der Wiederbesiedlungsquellen frühestens nach 5 Jahren zeigt. Mit einer deutlichen Annäherung ist eher ca. 10 Jahre nach der Renaturierung zu rechnen. Anhand der in der dritten Studie beschriebenen Veränderungswerte können realistische Ziele für Gewässerrenaturierungen gesetzt werden und zudem kann frühzeitig erkannt werden, ob die wasserwirtschaftlichen Ziele, bzw. das Erreichen des Guten Ökologischen Zustands für ein Gewässer überhaupt realistisch ist oder nicht.

Oftmals bildet sich der Erfolg nach einer Renaturierungmaßnahme jedoch nicht im Guten Ökologischen Potenzial ab. Trotzdem ist es wichtig zu wissen, ob nach einer Renaturierungsmaßnahme bereits ein „kleiner Erfolg“ stattgefunden hat und der renaturierte Abschnitt in Richtung einer taxonomischen Verbesserung verläuft. Hierzu können die Ausbreitungsklassen aus der zweiten Studie verwendet werden. Die Ausbreitungsklassen sind auf andere Einzugsgebiete übertragbar und zusätzlich erfasste Taxa können leicht anhand der beschrieben Regeln nachklassifiziert werden. Die Ausbreitungsklassen können als ein generelles Merkmal für die Analyse des Reifezustands einer Zönose verwendet werden, da sie konstantere Werte liefern als beispielsweise Taxazahlen. Kleinere Erfolge, die über die Bewertung des Guten Ökologischen Potenzials noch nicht abgebildet werden, wie beispielsweise die Wiederansiedlung anspruchsvoller Arten, werden so ersichtlich. Um jedoch den Wiederbesiedlungsprozess und die Sukzession noch besser zu verstehen, sollten zukünftig Langzeitstudien und idealerweise auch populationsgenetische Untersuchungen durchgeführt werden.

References

Beavan L., Sadler J. & Pinder C. (2001) The invertebrate fauna of a physically modified urban river. Hydrobiologia, 445, 97–108.

Becker A. & Robson B.J. (2009) Riverine macroinvertebrate assemblages up to eight years after riparian restoration in a semi-rural catchment in Victoria, Australia. Marine & Freshwater Research, 60, 1309–1316.

Beisel J.N., Usseglio-Polatera P. & Moreteau J.C. (2000) The spatial heterogeneity of a river bottom: a key factor determining macroinvertebrate communities. Hydrobiologia, 422/423, 163–171.

Bell N., Riis T., Suren A.M. & Baattrup-Pedersen A. (2013) Distribution of invertebrates within beds of two morphologically contrasting stream macrophyte species. Fundamental and Applied Limnology, 183, 309−321.

Benjamini Y. & Hochberg Y. (1995) Controlling the false discovery rate: a practical and powerful approach to multiple testing. Journal of the Royal Statistical Society, 57, 289–300.

Bernardt E.S. & Palmer M.A. (2007) Restoring streams in an urbanizing world. Freshwater Biology, 52, 738−751.

Bilton D.T., Freeland J.R. & Okamura B. (2001) Dispersal in freshwater invertebrates. Annual Review of Ecology and Systematics, 32, 159−181.

Bis B. & Usseglio-Polatera P. (2004) STAR Deliverable N2. Species Traits Analysis. Available at: www.eu-star.at/pdf/Deliverable_N2.pdf.

Black R.W., Munn M.D. & Plotnikoff R.W. (2004) Using macroinvertebrates to identify biota-land cover optima at multiple scales in the Pacific Northwest, USA. Journal of the North American Benthological Scociety, 23, 340–62.

Blakely J.T., Harding J.S., McIntosh A.R. & Winterbourn M.J. (2006) Barriers to the recovery of aquatic insect communities in urban streams. Freshwater Biology, 51, 1634−1645.

Blann K., Anderson J. L., Sands G. & Vondracek B. (2009) Effects of agricultural drainage on aquatic ecosystems: a Review. Critical Reviews in Environmental Science and Technology, 39, 909–1001.

Boix D., Sala J., Quintana X.D. & Moreno-Amich R. (2004) Succession of the animal community in a Mediterranean temporary pond. Journal of the North American Benthological Society, 23, 29–49.

Boix D., Biggs J., Céréghino R., Hull A.P., Kalettka T. & Oertli, B. (2012) Pond research and management in Europe: "Small is Beautiful". Hydrobiologia, 689, 1–9.

Borja A., Muxika I. & Franco J. (2006) Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervión estuary (southern Bay of Biscay). Marine Ecology Progress Series, 313, 43–55.

Boulton A.J., Findlay S., Marmonier P., Stanley E.H. & Valett H.M. (1998) The functional significance of the hyporheic zone in streams and rivers. Annual Review of Ecology and Systematics, 29, 59–81.

Braak C.J.F. ter & Smilauer P. (2003) CANOCO for Windows version 4.51. Biometris-Plant Research International, Wageningen.

Briers R.A., Cariss H.M. & Gee J.H.R. (2003) Flight activity of adult stoneflies in relation to weather. Ecological Entomology, 28, 31–40.

Cañedo-Argüelles M. & Rieradevall M. (2011) Early succession of the macroinvertebrate community in a shallow lake: Response to changes in the habitat condition. Limnologica, 41, 363–370.

Cañedo-Argüelles M., Boersma K.S., Bogan M.T., Olden J.D., Phillipsen I., Schriever T.A. & Lytle D.A. (2015) Dispersal strength determines meta-community structure in a dendritic riverine network. Journal of Biogeography, 42, 778–790.

Carpenter S.R. & Lodge D.M. (1986) Effects of submerged macrophytes on ecosystem processes. Aquatic Botany, 26, 341−370.

Clarke K.R. (1993) Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, 18, 117–143.

Coles J.F., McMahon G., Bell A.H., Brown L.R., Fitzpatrick F.A., Scudder-Eikenberry B.C., Woodside M.D., Cuffney T.F., Bryant W.L., Cappiella K., Fraley-McNeal L. & Stack W.P. (2012) Effects of urban development on stream ecosystems in nine metropolitan study areas across the United States. U.S. Geological Survey Circular, 1373, 138 p. Available at: http://pubs.usgs.gov/circ/1373/.

Collier K.J., Rutherford J.C., Quinn J.M. & Davies-Colley R.J. (2001) Forecasting rehabilitation outcomes for degraded New Zealand pastoral streams. Water Science and Technology, 43, 175–184.

Connell J.H. (1978) Diversity in tropical rain forests and coral reefs. Science, 199, 1302–1310.

Cushing C. & Gains W. (1989) Thoughts on recolonization of endorheic cold desert spring-streams. Journal of the North American Benthological Society, 8, 277–287.

Death R.G. & Collier K.J. (2010) Measuring stream macroinvertebrate responses to gradients of vegetation cover: When is enough enough? Freshwater Biology, 55, 1447–1464.

DIN 38410-1 (1987) Deutsche Einheitsverfahren zur Wasser-, Abwasser- und Schlammuntersuchung - Biologisch-ökologische Gewässeruntersuchung (Gruppe M) - Teil 1: Bestimmung des Saprobienindex in Fließgewässern (M 1).

EGLV (Emschergenossenschaft/Lippeverband) (2015) Generationenprojekt für eine Region mit Zukunft. Available at: www.eglv.de/wasserportal/emscher-umbau.html.

European Commission (2000) Directive 2000/60/EEC, Establishing a framework for community action in the field of water policy. Official Journal of the European Communities L327, 1−71, Brussels, Belgium.

Feld C.K. & Hering D. (2007) Community structure of function: effects of environmental stress on benthic macroinvertebrates at different spatial scales. Freshwater Biology, 52, 1380–99.

Feld C.K., Birk S., Bradley D.C., Hering D., Kail J., Marzin A., et al. (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Advances in Ecological Research, 44, 119−209.

Friberg N., Kronvang B., Hansen H.O. & Svendsen L.M. (1998) Long-term, habitat-specific response of a macroinvertebrate community to river restoration. Aquatic Conservation: Marine and Freshwater Ecosystems, 8, 87–99.

Gellert G., Behrens S. & Raschke M. (2012) The return of degraded stream ecosystems by using positive impacts from near-natural sections: A new practical guide for restorations. Water and Environment Journal, 26, 415−421.

Gellert G., Pottgiesser T. & Euler T. (2014) Assessment of the structural quality of streams in Germany--basic description and current status. Environmental Monitoring and Assessment, 186, 3365−3378.

Gerhard M. & Reich M. (2000) Restoration of streams with large wood: effects of accumulated and built-in wood on channel morphology, habitat diversity and aquatic fauna. International Review of Hydrobiology, 85, 123–137.

Glöer P. & Meier-Brook M. (2003) Süßwassermollusken. Ein Bestimmungsschlüssel für die Bundesrepublik Deutschland. Verlag Deutscher Jugendbund für Naturbeobachtungen, Hamburg.

Gore J.A. (1982) Benthic invertebrate colonization: source distance effects on community composition. Hydrobiologia, 9, 183–193.

Gregg W.W. & Rose F.L. (1982) The effects of aquatic macrophytes on the stream microenvironment. Aquatic Botany, 14, 309−324.

Gustafsson S., Österling M., Skurdal J., Schneider L.D. & Calles O. (2013) Macroinvertebrate colonization of a nature-like fishway: The effects of adding habitat heterogeneity. Ecological Engineering, 61, 345–353.

Haase P., Lohse S., Pauls S., Schindehütte K., Sundermann A., Rolauffs P. & Hering D. (2004) Assessing streams in Germany with benthic invertebrates: development of a practical standardised protocol for macroinvertebrate sampling and sorting. Limnologica, 34, 349–365.

Haase P., Murray-Bligh J., Lohse S., Pauls S.U., Sundermann A., Gunn R. & Clarke R. (2006) Assessing the impact of errors in sorting and identifying macroinvertebrate samples. Hydrobiologia, 566, 505–521.

Haase P., Sundermann, A. & Schindehütte K. (2011) Operationelle Taxaliste als Mindestanforderung an die Bestimmung von Makrozoobenthosproben aus Fließgewässern zur Umsetzung der EU-Wasserrahmenrichtlinie in Deutschland. Available at: www.fliessgewaesserbewertung.de/downloads/Operationelle_Taxaliste_Mai2011.zip.

Haase P., Hering D., Jähnig S.C., Lorenz A.W. & Sundermann A. (2013) The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates, and macrophytes. Hydrobiologia, 704, 475−488.

Heatherly T., Whiles M.R., Gibson D.J., Collins S.L., Huryn A.D., Jackson J.K. & Palmer M.A. (2007) Stream insect occupancy-frequency patterns and metapopulation structure. Oecologia, 151, 313–321.

Hering D., Moog O., Sandin L. & Verdonschot P.F.M. (2004) Overview and application of the AQEM assessment system. Hydrobiologia, 516, 1–20.

Hering D., Johnson R.K., Kramm S., Schmutz S., Szoszkiewicz K. & Verdonschot P.F.M. (2006) Assessment of European rivers with diatoms, macrophytes, invertebrates and fish: A comparative metric-based analysis of organism response to stress. Freshwater Biology, 51, 1757−1785.

Hering D., Borja A., Carstensen J., Carvalho L., Elliott M., Feld C.K., Heiskanen A.-S., Johnson R.K., Moe J., Pont D., Lyche Solheim A. & van de Bund W. (2010) The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. Science of the Total Environment, 408, 4007−4019.

Huang S. & Guo Q. (2014) Research review on effects of urban landscape pattern changes on water environment. Shengtai Xuebao/Acta Ecologica Sinica, 34, 3142–3150.

Hughes J.M. (2007) Constraints on recovery: using molecular methods to study connectivity of aquatic biota in rivers and streams. Freshwater Biology, 52, 616–631.

Hughes S.J., Ferreira T., Cortes R.V. (2008) Hierarchical spatial patterns and drivers of change in bentic macroinvertebrate communities in an intermittent Mediterranean river. Aquatic Conservation: Marine and Freshwater Ecosystems, 18, 742–60.

Hynes H.B.N. (1955) The Reproductive Circle of Some British Freshwater Gammaridae. Journal of Animal Ecology, 24, 352−387.

Jähnig S.C., Lorenz A.W. & Hering D. (2009) Habitat mosaics and macroinvertebrates - does channel form determine community composition? Aquatic Conservation: Marine and Freshwater Ecosystems, 19, 157−169.

Jähnig S.C., Brabec K., Buffagni A., Erba S., Lorenz A.W., Ofenböck T., Verdonschot P.F.M. & Hering D. (2010) A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology, 47, 671−680.

Jones H.P. & Schmitz O.J. (2009) Rapid Recovery of Damaged Ecosystems. PLoS ONE, 4, e5653.

Jones K.B., Neale A.C., Nash M.S., Van Remortel R.D., Wickham J.D., Riitters K.H. & O'neill R.V. (2001) Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology, 16, 301–312.

Jungwirth M., Moog O. & Muhar S. (1993) Effects of river bed restructuring on fish and benthos of a 5th-order stream, Melk, Austria. Regulated Rivers: Research & Management, 8, 195–204.

Kail J. & Hering D. (2005) Using large wood to restore streams in Central Europe: potential use and likely effects. Landscape Ecology, 20, 755–772.

Kail J. & Hering D. (2009) The influence of adjacent stream reaches on the local ecological status of Central European mountain streams. River Research and Applications, 25, 537–550.

Kail J. & Wolter C. (2013) Pressures at larger spatial scales strongly influence the ecological status of heavily modified river water bodies in Germany. Science of the Total Environment, 454-455, 40−50.

Kenney M.A., Wilcock P.R., Hobbs B.F., Flores N.E. & Martínez D.C. (2012) Is urban stream restoration worth it? Journal of the American Water Resources Association, 48, 603–615.

Kiffney P.M., Richardson J.S. & Bull J.P. (2003) Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. Journal of Applied Ecology, 40, 1060–1076.

Lake P.S., Bond N. & Reich, P. (2007) Linking ecological theory with stream restoration. Freshwater Biology, 52, 597–615.

Langford T.E.L., Shawa P.J., Ferguson A.J.D. & Howard S.R. (2009) Long-term recovery of macroinvertebrate biota in grossly polluted streams: Re-colonization as a constraint to ecological quality. Ecological Indicators, 9, 1064–1077.

LAWA (Länderarbeitsgemeinschaft Wasser) (2004) Abschlussbericht des Forschungsprojektes "Abschließende Arbeiten zur Fließgewässertypisierung entsprechend den Anforderungen der WRRL" (FKZ O 7.03).

LAWA (Länderarbeitsgemeinschaft Wasser) (2013) Bewertung von HMWB/AWB-Fließgewässern und Ableitung des HÖP/GÖP. final report. Hilden. Available at: http://www.laenderfinanzierungsprogramm.de/cms/WaBoAb_prod/WaBoAb/Vorhaben/

LAWA/Vorhaben_des_Ausschusses_Oberflaechengewaesser_und_Kuestengewaesser_%28AO%29/O_3.10/ENDBERICHT_O3-10_HMWB_FINAL_Juli_2013.pdf.

Lorenz A., Feld C.K. & Hering D. (2004) Typology of streams in Germany based on benthic invertebrates: Ecoregions, zonation, geology and substrate. Limnologica, 34, 390−397.

Lorenz A.W., Jähnig S.C. & Hering D. (2009) Re-meandering German lowland streams - qualitative and quantitative effects of restoration measures on hydromorphology and benthic invertebrates. Environmental Management, 44, 745–754.

Lorenz A.W. & Feld C.K. (2013) Upstream river morphology and riparian land use overrule local restoration effects on ecological status assessment. Hydrobiologia, 704, 489–501.

LUA NRW (Landesumweltamt Nordrhein-Westfalen) 1998. Gewässerstrukturgüte in Nordrhein-Westfalen. Merkblatt 14. Essen.

Malmqvist B. (2002) Aquatic invertebrates in riverine landscapes. Freshwater Biology, 47, 679–694.

Mann H.B. & Whitney D.R. (1947) On a test whether one of two random variables is stochastically larger than the other. Annals of Mathematical Statistics, 18, 50−60.

McCook L.J. (1994) Understanding ecological community succession: Causal models and theories, a review. Vegetatio, 110, 115–147.

Meier C., Haase P., Rolauffs P., Schindehütte K., Schöll F., Sundermann A. & Hering D. (2006) Methodisches Handbuch Fließgewässerbewertung. Handbuch zur untersuchung und Bewertung von Fließgewässern auf Basis des Makrozoobenthos vor dem Hintergrund der EG-Wasserrahmenrichtlinie. Available at: www.fliessgewaesserbewertung.de.

Miguel-Chinchilla L., Boix D., Gascón S. & Comín F.A. (2014) Macroinvertebrate biodiversity patterns during primary succession in manmade ponds in North-Eastern Spain. Journal of Limnology, 73, 428–440.

Milner A.M., Knudsen E.E., Soiseth C., Robertson A.L., Schell D., Phillips I.T. & Magnusson K. (2000) Colonization and development of stream communities across a 200-year gradient in Glacier Bay National Park, Alaska, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences, 57, 2319–2335.

Narf R.P. (1985) Aquatic insect colonization and substrate changes in a relocated stream segment. The Great Lakes Entomologist, 18, 83–92.

Nienhuis P.H., Buijse A.D., Leuven R.S.E.W., Smits A.J.M., De Nooij R.J.W. & Samborska E.M. (2002) Ecological rehabilitation of the lowland basin of the river Rhine (NW Europe). Hydrobiologia, 478, 53–72.

Nijboer R.C. & Schmidt-Kloiber A. (2004) The effect of excluding taxa with low abundances or taxa with small distribution ranges on ecological assessment. Hydrobiologia, 516, 349−366.

Palmer M.A., Menninger H.L. & Bernhardt E. (2010) River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biology, 55, 205−222.

Palmer M.A., Hondula K.L. & Koch B.J. (2014) Ecological restoration of streams and rivers: shifting strategies and shifting goals. Annual Review of Ecology and Systematics, 45, 247–269.

Pander J. & Geist J. (2013) Ecological indicators for stream restoration success. Ecological Indicators, 30, 106–118.

Parkyn S.M., Davies-Colley R.J., Halliday N.J., Costley K.J. & Croker G.F. (2003) Planted riparian buffer zones in New Zealand: Do they live up to expectations? Restoration Ecology, 11, 436–447.

Parkyn S.M. & Smith B.J. (2011) Dispersal constraints for stream invertebrates: Setting realistic timescales for biodiversity restoration. Environmental Management, 48, 602–614.

Paul M.J. & Meyer J.L. (2001) Streams in the urban landscape. Annual Review of Ecology and Systematics, 32, 333–365.

Prange H. (2005) Die ökologische Bedeutung der Verockerung und ihre Relevanz für die Umsetzung der EG-WRRL. unpublished report. Bremen.

Renöfält B., Nilsson C., Jansson R. (2005) Spatial and temporal patterns of species richness in a riparian landscape. Journal of Biogeography, 32, 2025−2037.

Reynolds L.K., Waycott M. & Mcglathery K.J. (2013) Restoration recovers population structure and landscape genetic connectivity in a dispersal-limited ecosystem. Journal of Ecology, 101, 1288–1297.

Ruhí A., Boix D., Sala J., Gascón S. & Quintana X.D. (2009) Spatial and temporal patterns of pioneer macrofauna in recently created ponds: Taxonomic and functional approaches. Hydrobiologia, 634, 137–151.

Ruhí A., Boix D., Gascón S., Sala J. & Quintana X.D. (2013) Nestedness and successional trajectories of macroinvertebrate assemblages in man-made wetlands. Oecologia, 171, 545–556.

Schmedtje U. & Colling M. (1996) Ökologische Typisierung der aquatischen Makrofauna. Bayerisches Landesamt für Wasserwirtschaft, München.

Schmidt-Kloiber A. & Hering D. (2012) www.freshwaterecology.info - the taxa and autecology database for freshwater organisms. Available at: www.freshwaterecology.info (Version 5.0).

Shields F.D., Knight S.S. & Cooper C.M. (1995) Use of the index of biotic integrity to assess physical habitat degradation in warm water streams. Hydrobiologia, 312, 191–208.

Spänhoff B. & Arle J. (2007) Setting Attainable Goals of Stream Habitat Restoration from a Macroinvertebrate View. Restoration Ecology, 15, 317−320.

Sundermann A., Stoll S. & Haase P. (2011a) River restoration success depends on the species pool of the immediate surroundings. Ecological Applications, 21, 1962–1971.

Sundermann A., Antons C., Cron N., Lorenz A.W., Hering D. & Haase P. (2011b) Hydromorphological restoration of running waters: effects on benthic invertebrate assemblages. Freshwater Biology, 56, 1689−1702.

Suren A.M. (2000) Effects of Urbanisation. In: New Zealand Stream Invertebrates: Ecology and Implications for Management (Eds. Collier K. J. & Winterbourn M. J.), pp. 260–288. New Zealand Limnological Society, Christchurch, New Zealand.

Tachet H., Richoux P., Bournaud M. & Usseglio-Polatera P. (2010) Invertébrés d´eau douce. systématique, biologie, écoloogie. CNRS Editions, Paris.

Tonkin J.D., Stoll S., Sundermann A. & Haase P. (2014) Dispersal distance and the pool of taxa, but not barriers, determine the colonisation of restored river reaches by benthic invertebrates. Freshwater Biology, 59, 1843–1855.

Tullos D.D., Penrose D.L., Jennings G.D. & Cope W.G. (2009) Analysis of functional traits in reconfigured channels: implications for the bioassessment and disturbance of river restoration. Journal of the North American Benthological Society, 28, 80–92.

Van Den Brink F.W.B., Van Der Velde G., Buijse A.D., Klink A.G. (1996) Biodiversity in the Lower Rhine and Meuse river-floodplains: Its significance for ecological river management. Aquatic Ecology, 30, 129–149.

Van Leeuwen C.H.A., Huig N., Van der Velde G., Van Alen T., Wagemaker C.A.M., Sheerman C.D.H., et al. (2013) How did this snail get here? Several dispersal vectors inferred for an aquatic invasive species. Freshwater Biology, 58, 88−99.

Verdonschot P.F.M. (2009) Impact of hydromorphology and spatial scale on macroinvertebrate assembalge composition in streams. Integrated Environmental Assessment and Management, 5, 97−109.

Vieira N.K.M, Poff N.L., Carlisle D.M., Moulton II S.R., Koski M.L. & Kondratieff B.C. (2006) A database of lotic invertebrate traits for North America: U.S. Geological Survey Data Series 187. Available at: http://pubs.water.usgs.gov/ds187-Vieira_2006_InvertTraits Table_v1.txt.

Violin C.R., Cada P., Sudduth E.B., Hassett B.A., Penrose D.L. & Bernhardt E.S. (2011) Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications, 21, 1932–1949.

Von Bertrab M., Krein A., Stendera S., Thielen F. & Hering D. (2013) Is fine sediment deposition a main driver for the composition of benthic macroinvertebrate assemblages? Ecological Indicators, 24, 589–598.

Walsh C.J., Roy A.H., Feminella J.W., Cottingham P.D., Groffman P.M. & Raymond P.M. II. (2005) The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society, 24, 706–723.

Winking C., Korte T. & Lorenz A.W. (2013) Die Wiederbesiedlung urbaner Fließgewässer in einem Teileinzugsgebiet der Emscher nach erfolgten ökologischen Verbesserungen. Korrespondenz Wasserwirtschaft 6, 310−317.

Winking, C., Lorenz, A.W., Sures, B. & Hering, D. (2014) Recolonisation patterns of benthic invertebrates: A field investigation of restored former sewage channels. Freshwater Biology, 59, 1932–1944.

Acknowledgements

First of all, I want to express deep gratitude to Prof. Dr. Daniel Hering for his brilliant supervision and the opportunity to work on such an interesting topic. I really appreciate our numerous discussions, his feedback and the constructive comments on my work.
I want to address special thanks to Dr. Armin Lorenz for the overall support, the fruitful discussions, the exchange of ideas and the comments on the thesis manuscript. He always knew how to cheer me up and helped me also to improve my own work to the highest.
This thesis is a result of the project KuLaRuhr as part of the German Federal Ministry of Education and Research program "Sustainable land use" (BMBF, project number: 033L020A, www.kularuhr.de). I want to say "thank you" to all my KuLaRuhr colleges for the teamwork and the great project time. My thanks go especially to my office colleges Daniel Dangel and Simon Kresman and further to Prof. Dr. Bernd Sures as the head of the project.
The whole working group of the Department of Applied Ecology is not only thanked for the helpful advices, suggestions and discussions, but also for the fantastic working atmosphere and the good time we had during and after work.
I thank Martin Sondermann for helping with calculating the land use patterns and Dr. Christian Feld and Dr. Jochem Kail for statistical advices. I am also thankful to Michael Hohenadler, Simon Kresmann, Hannah Jachertz, Alexander Gieswein and Maria Gies for helpful comments and remarks on the dissertation manuscript or the paper manuscripts.
I grateful acknowledge Claudia Schirmer, Louisa Rothe, Hanna Lücking and Katja Beyer for the great time, the fun and the assistance throughout many field trips and the lab work. Special thanks go to Family Rothe and their motorbus, although it broke down during a field trip.
I would like to thank the Emschergenossenschaft/Lippeverband (Essen) for providing data and information of the Emscher catchment, and generally for their friendly help, and support.
Finally I am deeply grateful to my parents, my brother and sisters, Marlon, Christian, Larissa, my friends Anika, Verena, Eva, Kathrin and especially Peter for being so encouraging and interested in my topic. Mia and Annette, thank you for your care and love throughout my hard times.

Appendix

Appendix 1

Table A1 87 environmental parameters used for the PCA analysis including their vector length and a description, why some parameters with a vector length of <0.2 still were absent in the PCA. PHQ=Physical Habitat Quality.

Environmental Parameters	Presence in PCA	Vector Length	Description
Age of restoration (m)	√	0.7647	
Presence of iron ochre (yes/no)	√	-0.3412	
Length of the restored section (m)	no	≤ 0.2	
Occurrence of a sewage overflow upstream of a sampling site (yes/no)	√	-0.2298	
Connection to a near-natural upstream stretch or tributary (yes/no)	√	0.2934	
Stones and gravel on the stream bottom (%)	no	≤ 0.2	
Share of sand and clay on the stream bottom (%)	no	≤ 0.2	
Share of artificial substances on the stream bottom (%)	no	≤ 0.2	
Share of algae, macrophytes and living parts of terrestrial plants on the stream bottom (%)	√	-0.3258	
Share of organic matter on the stream bottom (%)	no	≤ 0.2	
Share of dead wood on the stream bottom (%)	√	0.386	
PHQ at the sampling site	√	-0.2033	
Mean PHQ 200m upstream the sampling site	no	≤ 0.2	similar to PHQ at the sampling site
Mean PHQ 1000m upstream the sampling site	no	≤ 0.2	similar to PHQ at the sampling site
Extensive built-up settlement area in a buffer of 20x100 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 20x100 m (%)	no	≤ 0.2	
Agricultural land in a buffer of 20x100 m (%)	no	≤ 0.2	
Grassland in a buffer of 20x100 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of	no	≤ 0.2	

Environmental Parameters	Presence in PCA	Vector Length	Description
20x100 m (%)			
Coniferous riparian vegetation in a buffer of 20x100 m (%)	no	≤ 0.2	
Surface waters in a buffer of 20x100 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 20x100 m (%)	no	≤ 0.2	
Extensive built-up settlement area in a buffer of 20x200 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 20x200 m (%)	no	≤ 0.2	
Agricultural land in a buffer of 20x200 m (%)	no	≤ 0.2	
Grassland in a buffer of 20x200 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 20x200 m (%)	no	≤ 0.2	
Coniferous riparian vegetation in a buffer of 20x200 m (%)	no	≤ 0.2	
Surface waters in a buffer of 20x200 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 20x200 m (%)	no	≤ 0.2	
Extensive built-up settlement area in a buffer of 20x500 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 20x500 m (%)	no	-0.2778	compare unsealed area (opposite)
Agricultural land in a buffer of 20x500 m (%)	no	≤ 0.2	
Grassland in a buffer of 20x500 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 20x500 m (%)	no	0.2966	similar to buffer 20x1000m
Coniferous riparian vegetation in a buffer of 20x500 m (%)	no	≤ 0.2	
Surface waters in a buffer of 20x500 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 20x500 m (%)	no	0.2778	similar to buffer 100x1000m
Extensive built-up settlement area in a buffer of 20x1000 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 20x1000 m (%)	no	-0.4813	compare unsealed area (opposite)
Agricultural land in a buffer of 20x1000 m (%)	no	≤ 0.2	

Environmental Parameters	Presence in PCA	Vector Length	Description
Grassland in a buffer of 20x1000 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 20x1000 m (%)	√	0.4126	
Coniferous riparian vegetation in a buffer of 20x1000 m (%)	no	≤ 0.2	
Surface waters in a buffer of 20x1000 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 20x1000 m (%)	no	0.4813	similar to buffer 100x1000m
Extensive built-up settlement area in a buffer of 100x100 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 100x100 m (%)	no	-0.2636	compare unsealed area (opposite)
Agricultural land in a buffer of 100x100 m (%)	no	≤ 0.2	
Grassland in a buffer of 100x100 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 100x100 m (%)	no	≤ 0.2	
Coniferous riparian vegetation in a buffer of 100x100 m (%)	no	≤ 0.2	
Surface waters in a buffer of 100x100 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 100x100 m (%)	no	0.2636	similar to buffer 100x1000m
Extensive built-up settlement area in a buffer of 100x200 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 100x200 m (%)	no	-0.2738	compare unsealed area (opposite)
Agricultural land in a buffer of 100x200 m (%)	no	≤ 0.2	
Grassland in a buffer of 100x200 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 100x200 m (%)	no	≤ 0.2	
Coniferous riparian vegetation in a buffer of 100x200 m (%)	no	≤ 0.2	
Surface waters in a buffer of 100x200 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 100x200 m (%)	no	0.2738	similar to buffer 100x1000m
Extensively built-up settlement area in a buffer of 100x500 m (%)	no	≤ 0.2	
Intensively built-up settlement area (= sealed area) in a buffer of 100x500 m (%)	no	-0.3597	compare unsealed area (opposite)

Environmental Parameters	Presence in PCA	Vector Length	Description
Agricultural land in a buffer of 100x500 m (%)	no	≤ 0.2	
Grassland in a buffer of 100x500 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 100x500 m (%)	no	0.2437	similar to buffer 20x1000m
Coniferous riparian vegetation in a buffer of 100x500 m (%)	no	≤ 0.2	
Surface waters in a buffer of 100x500 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 100x500 m (%)	no	0.3597	similar to buffer 100x1000m
Extensive built-up settlement area in a buffer of 100x1000 m (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in a buffer of 100x1000 m (%)	no	-0.5121	compare unsealed area (opposite)
Agricultural land in a buffer of 100x1000 m (%)	no	≤ 0.2	
Grassland in a buffer of 100x1000 m (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in a buffer of 100x1000 m (%)	no	0.2518	similar to buffer 20x1000m
Coniferous riparian vegetation in a buffer of 100x1000 m (%)	no	≤ 0.2	
Surface waters in a buffer of 100x1000 m (%)	no	≤ 0.2	
Unsealed area in a buffer of 100x1000 m (%)	√	0.5121	
Extensive built-up settlement area in the sub-catchment (%)	no	≤ 0.2	
Intensive built-up settlement area (= sealed area) in the sub-catchment (%)	√	≤ 0.2	in PCA for comparison to buffers
Agricultural land in the sub-catchment (%)	no	≤ 0.2	
Grassland in the sub-catchment (%)	no	≤ 0.2	
Deciduous and mixed riparian vegetation in the sub-catchment (%)	no	≤ 0.2	
Coniferous riparian vegetation in the sub-catchment (%)	no	≤ 0.2	
Surface waters in the sub-catchment (%)	no	≤ 0.2	
Unsealed area in the sub-catchment (%)	√	≤ 0.2	in PCA for comparison to buffers
Contaminated areas in the sub-catchment (%)	no	≤ 0.2	

Appendix 2

*Table A2 Categorisation of taxa according to dispersal capability and ecological preferences based on literature data. x = true value; 0 = no active areal dispersal, 1-3 = dispersal capability low, middle, high; column meanings as described in methods/dispersal classes; * = If the dispersal information derived from Bis & Usseglio-Polatera (2004) and/or Tachet et al. (2010) is given only at genus level, it was transferred to species level; pot. pre. = potential presence; SI = saprobic index, FR_RC =frequency in restored, connected sites, FR_RU = frequency in restored unconnected sites, FR_sum = frequency in all restored sites, FR_SB, SO = frequency in source sites, RQ = recolonisation quotient.*

References:

1 Amann et al. (1994); 2 Angelibert & Giani (2003); 3 Askew (1988); 4 Bagge (1995); 5 Bauernfeind (1990); 6 Bayerisches Landesamt für Wasserwirtschaft/Schmedtje & Colling (1996); 7 Bellmann (1987); 8 Bellmann (1988); 9 Bis & Usseglio-Polatera (2004); 10 Blanke (1990); 11 Botosaneanu & Malicky (1978); 12 Braasch & Jakob (1976); 13 Brandstetter & Kapp (1995); 14 Brauckmann (1987); 15 Brehm & Meijering (1990); 16 Brockhaus (1991); 17 Burmeister (1984); 18 Burmeister (1986); 19 Burmeister & Reiss (1983); 20 Bussler (1992); 21 Caspers (1980); 22 Caspers & Stiers (1977); 23 Chaput-Bardy et al. (2008); 24 Chaput-Bardy et al. (2010); 25 Chinery (1987); 26 Conrad et al. (1999); 27 Conze et al. (2011); 28 Dedecker et al. (2004); 29 Delette & Morvan (2000); 30 Deutscher Rat für Landespflege (2009); 31 Donath (1989); 32 Dreyer (1986); 33 Ehlert (2009); 34 Elliott (2008); 35 Engelhardt (1986); 36 Göthberg (1973); 37 Graf et al. (1995); 38 Havelka (1992); 39 Hebauer (1974); 40 Heidemann & Seidenbusch (1993); 41 Holm (1989); 42 Horion (1960); 43 Illies & Botosaneanu (1963); 44 Jacobs & Renner (1988); 45 Janecek & Contreras (1995); 46 Kehl & Dettner (2007); 47 Klausnitzer (1971); 48 Klausnitzer (1984); 49 Klausnitzer et al. (1978); 50 Klein (1984); 51 Klima (1994); 52 Koch (1989); 53 Kuusela & Huusko (1996); 54 Lehmann (1971); 55 Ludwig (1989); 56 Madsen et al. (1973); 57 Maibach & Meier (1987); 58 Malicky (1987); 59 Malzacher (1981); 60 Masters et al. (2007); 61 Mey (1993); 62 Meyer (1987); 63 Moog (1995); 64 Müller-Liebenau (1969); 65 Petersen et al. (1999); 66 Pitsch & Weinzierl (1992); 67 Reiff (1993); 68 Reiter (1993); 69 Samietz (1989); 70 Sauer (1987); 71 Sauer (1988); 72 Schmedtje (1995); 73 Schmedtje & Kohmann (1992); 74 Schmedtje & Zwick (1992); 75 Schmidt-Kloiber & Hering (2012); 76 Schorr (1990); 77 Schutte et al. (1997); 78 Seitz & Weinzierl (1991); 79 Sode & Wiberg-Larsen (1993); 80 Stettmer (1996); 81 Svensson (1974); 82 Tachet et al. (2010); 83 Thiele et al. (1998); 84 Thomes (1987); 85 Timm (1993); 86 Tobias & Tobias (1981); 87 Van Noordwijk (1978); 88 Vieira et al. (2006); 89 Wagner (1978); 90 Wagner (1992); 91 Wagner & Gerecke (2008); 92 Ward & Mill (2007); 93 Waringer (1986); 94 Waringer (1989); 95 Weinzierl & Dorn (1989); 96 Zahner (1960).

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
7381	*Anodonta anatina*				0					6;9;75	**A**	2	0	0	0	2	0	-
8691	*Asellus aquaticus*				0					6;9;75	**A**	2.8	13	9	22	18	1.25	-
4462	*Bithynia tentaculata*				0				x	6;9;75	**A**	2.3	2	0	2	0	-	-
4911	*Dendrocoelum lacteum*				0					6;9;75	**A**	2.4	2	0	2	0	-	-
4973	*Dina lineata*				0		x		x	6;9;75	**A**	-	0	0	0	2	0	-
5018	*Dugesia gonocephala*				0	x				6;9;75	**A**	1.5	0	0	0	11	0	-
5075	*Eiseniella tetraedra*				0					6;9;75	**A**		0	0	0	13	0	-
5101	Enchytraeidae gen. sp.				0					6;9;75	**A**	-	0	0	0	2	0	-
5159	*Erpobdella octoculata*				0					6;9;75	**A**	2.8	2	2	4	22	0.2	-
5157	*Erpobdella vilnensis*				0					6;9;75	**A**	2.2	0	0	0	4	0	-
5284	*Galba truncatula*				0		x			6;9;75	**A**	2.1	0	0	0	2	0	-
5288	*Gammarus fossarum*				0	x				6;9;75	**A**	1.5	2	4	7	16	0.43	-
5291	*Gammarus pulex*				0	x				6;9;75	**A**	2	11	4	16	60	0.26	-
5304	*Glossiphonia complanata*				0					6;9;75	**A**	2.3	2	2	4	33	0.13	-
5354	*Gyraulus albus*				0		x			6;9;75	**A**	2	0	0	0	2	0	-
5373	*Haemopis sanguisuga*				0		x			6;9;75	**A**		0	0	0	2	0	-
5413	*Helobdella stagnalis*				0				x	6;9;75	**A**	2.6	0	0	0	13	0	-
5900	Lumbricidae gen. sp.				0					6;9;75	**A**	-	7	4	11	40	0.28	-
5907	*Lumbriculus variegatus*				0		x			6;9;75	**A**	3	0	0	0	9	0	-
7966	*Musculium lacustre*				0		x			6;9;75	**A**	2	0	0	0	4	0	-
20200	Naididae/Tubificidae gen. sp.				0					6;9;75	**A**	-	7	2	9	36	0.25	-
6127	*Niphargus* sp.			x	0					6;9;75	**A**	-	0	0	0	2	0	-

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
6395	*Physa fontinalis*				0		x			6;9;75	**A**	3.6	7	2	9	0	-	-
6425	*Pisidium* sp.				0					6;9;75	**A**	-	7	2	9	44	0.2	-
6430	*Planaria torva*				0		x			6;9;75	**A**	2.3	0	0	0	2	0	-
6436	*Planorbis planorbis*				0		x		x	6;9;75	**A**	2.4	2	2	4	2	2	-
8251	*Potamopyrgus antipodarum*				0					6;9;75	**A**	2.3	4	0	4	16	0.29	-
8703	*Proasellus coxalis*				0	x				6;9;75	**A**	2.8	0	2	2	13	0.17	-
16959	*Radix balthica*				0					6;9;75	**A**	2.3	11	11	22	22	1	-
6882	*Sphaerium corneum*				0		x		x	6;9;75	**A**	2,4	4	0	4	2	2	-
6905	*Stagnicola palustris*				0		x			6;9;75	**A**	2	4	0	4	4	1	-
6935	*Stylodrilus heringianus*				0					6;9;75	**A**	-	0	0	0	4	0	-
4293	*Amphinemura* sp.	x	x		1					9;15;72;82	**B**	1.5	0	0	0	4	0	no statement possible
20196	*Atherix/Ibisia* sp.	x	x		1	x				9;14;17;74;75;82;85;88	**B**	2	0	0	0	2	0	no statement possible
17788	*Elodes minuta*-gr.	x		x	1					6;8;9;21;41;47;75;82	**B**	1.5	4	0	4	44	0.1	yes (only connected sites)
5442	*Hemerodromia* sp.	x	x		1	x				9;75;82;90	**B**	2	0	0	0	2	0	no statement possible
6095	*Nemoura cinerea cinerea*	x		x	1		x			9;22;53;65;75;82;95	**B**	1.5	13	0	13	40	0.33	yes (only connected sites)
9747	*Sericostoma flavicorne/personatum*	x		x	1					9;75;79;82	**B**	1.5	0	0	0	2	0	no statement possible
4300	*Anabolia nervosa*	x	x		3*					6;9;75;82	**C**	2	4	0	4	0	-	no statement possible (flight > 5km ?)
17503	*Anacaena globulus*	x		x	3*					9;75;82	**C**	2	0	0	0	4	0	no statement possible
4638	*Chelifera* sp.	x	x	x	3					91	**C**	0	0	0	0	9	0	yes
4740	*Cordulegaster boltonii*	x		x	3	x				3;16;31;32;41;45;75	**C**	1.5	0	0	0	11	0	yes
5318	*Glyphotaelius pellucidus*	x	x		3		x	x		6;9;75;82	**C**	0	7	0	7	20	0.33	yes (only connected sites)

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
5376	*Halesus radiatus*	x	x		3					25;75;82	**C**	1.9	2	0	2	7	0.33	yes (only connected sites)
5894	*Lithax obscurus*	x	x	x	3*	x				6;19;37;41;51;75;86;82	**C**	1.5	0	0	0	2	0	no statement possible
5921	*Lype reducta*	x	x		3					37;75	**C**	0	0	0	0	7	0	yes
14488	*Micropterna lateralis/sequax*	x	x	x	3					6;21;37;51;75;82;86	**C**	-	4	0	4	18	0.25	yes (only connected sites)
6022	*Micropterna nycterobia*	x	x	x	3					6;21;51;75;82;87	**C**	-	0	0	0	4	0	no statement possible
6168	*Odontocerum albicorne*	x	x	x	3	x				41;75;82	**C**	1.4	0	0	0	2	0	no statement possible
6444	*Plectrocnemia conspersa conspersa*	x		x	3					8;11;21;41;51;61;75;82	**C**	1.5	0	0	0	27	0	yes
6524	*Potamophylax nigricornis*	x	x	x	3					6;19;21;37;51;75;82;86	**C**	1	2	0	2	2	1	yes (only connected sites)
6526	*Potamophylax rotundipennis*	x	x		3			x		21;75;82	**C**	2	2	0	2	0	-	no statement possible (flight > 5km ?)
6527	*Potamophylax* sp.	x	x		3					21;36;79;81;82	**C**	2	4	0	4	7	0.67	yes (only connected sites)
21224	*Tinodes waeneri waeneri*	x	x		2*		x			9;51;75;82;86	**C**	2	0	2	2	0	-	no statement possible (flight > 5km ?)
14467	*Chrysopilus* sp.	x			1					17	**D**	0	0	0	0	4	0	no statement possible
9324	*Chrysops* sp.	x			1		x			17;75	**D**	0	0	0	0	20	0	yes
4955	*Dicranota* sp.	x			1*					82;88	**D**	0	0	2	2	18	0.13	yes (only old unconnected sites)
4989	*Dixa* sp.	x			1					9;82	**D**	0	0	0	0	4	0	no statement possible
10349	*Dixella* sp.	x			1		x			9;75;82	**D**	0	0	2	2	2	1	no statement possible
20169	*Elmis aenea/maugetii*	x			1	x				9;34;75;82	**D**	2	2	0	2	7	0.33	yes
9654	*Eloeophila* sp.	x			1*					82	**D**	0	0	2	2	22	0,1	yes (only old unconnected sites)
9599	*Ephydridae* gen. sp.	x			1					6;9;44;49;82	**D**	0	0	2	2	2	1	no statement possible

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
5657	*Ironoquia dubia*	x			1		x			9;37;63;66;75;82;86	**D**	2	2	0	2	7	0.33	yes
18421	*Limnius volckmari*	x			1*	x				9;75;82	**D**	1.6	2	0	2	7	0.34	yes
6583	*Prodiamesa olivacea*	x			3					54;69;82	**D**	0	11	0	11	36	0.31	no --> change from E to D
7259	*Pseudolimnophila* sp.	x			1					67;82;88	**D**	-	0	0	0	11	0	yes
8753	*Psychodidae* gen. sp.	x			1					9;70;82	**D**	0	0	0	0	18	0	yes
7492	*Ptychoptera* sp.	x			1					9;82;89	**D**	0	4	0	4	31	0.14	yes
6822	*Sialis lutaria*	x			1*		x			9;14;75;82	**D**	2.5	4	4	9	11	0.8	yes (only old unconnected sites)
6853	*Simulium* sp.	x			2	x				6;55;75;82	**D**	0	18	0	18	22	0.7	no --> change from E to D
17473	*Agabus didymus*	x			3	x				42;46;48;75;82	**E**	2	4	7	11	4	2.5	yes
11656	*Agabus paludosus*	x			3					6;46;48;52;75;82	**E**	-	0	0	0	4	0	no statement possible
17492	*Agabus* sp.	x			3					6;46;68;75;82	**E**	0	2	2	4	9	0.5	yes
4260	*Agrypnia varia*	x			3*		x			9;72;75;82;86	**E**	0	0	4	4	0	-	no statement possible (flight > 5km ?)
17504	*Anacaena limbata*	x			3*		x			9;75;82	**E**	0	0	0	0	4	0	no statement possible
4415	*Baetis rhodani*	x			3					4;5;8;28;56;60;64;67;82;88	**E**	2.1	2	2	4	13	0.33	yes
4528	*Caenis* sp.	x			2		x			8;9;35;44;56;75;82;87	**E**	2	0	2	2	0	-	no statement possible (flight > 5km ?)
4530	*Calopteryx splendens*	x			3				x	6;7;8;23;24;27;30;41;75;77;80 82;88;92;94;96	**E**	2.2	2	2	4	2	2	yes
4585	*Ceratopogonidae* gen. sp.	x			3					9;38	**E**	0	2	7	9	47	0.19	yes
4642	*Chironomidae* gen. sp.	x			3					8;29;49	**E**	0	16	13	29	69	0.42	yes
4644	*Chironomini* gen. sp.	x			3					8;9;49;82	**E**	0	7	4	11	33	0.33	yes
10897	*Chironomus riparius*-agg.	x			3*					8;9;49;82	**E**	0	0	0	0	2	0	no statement possible

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
10900	*Chironomus thummi*-gr.	x			3*					8;9;49;82	**E**	0	2	0	2	2	1	no statement possible
4705	*Cloeon dipterum*	x			3*		x			5;8;9;12;59;75;82	**E**	2.3	4	9	13	4	3	yes
11165	*Coenagrion puella/pulchellum*	x			3		x			2;3;7;57;27;75;76;82	**E**	0	0	2	2	0	-	no statement possible (flight > 5km ?)
4723	Coenagrionidae gen. sp.	x			3		x			75;88	**E**	0	4	9	13	2	6	yes
11723	Colymbetinae gen. sp.	x			3					s. *Agabus* (Dytiscidae)	**E**	-	0	4	4	2	2	yes
7726	Culicidae gen. sp.	x			3		x			75;82;88	**E**	0	0	0	0	2	0	no statement possible
17684	*Cyphon* sp.	x			3		x			9;75;82	**E**	0	0	0	0	4	0	no statement possible
17749	*Dryops* sp.	x			3		x			9;75;82	**E**	0	0	0	0	4	0	no statement possible
5124	*Ephemera danica*	x			3					28;33;82	**E**	1.8	2	0	2	4	0.5	no statement possible
17901	*Haliplus* sp.	x			3		x			1;8;9;13;43;48;75;82	**E**	0	4	4	9	4	2.22	yes
18157	*Hydrobius fuscipes*	x			3		x			6;9;75;82	**E**	0	0	0	0	2	0	no statement possible
18251	*Hydroporus* sp.	x			3					9;46;82	**E**	0	0	2	2	0	-	no statement possible (flight > 5km ?)
5588	*Hydropsyche angustipennis angustipennis*	x			3			x		9;37;51;61;67;75;82;83	**E**	2.3	4	7	11	11	1	yes
18321	*Ilybius* sp.	x			3		x			9;75;82	**E**	0	0	0	0	2	0	no statement possible
5658	*Ischnura elegans*	x			3		x		x	3;7;26;27;40;50;55;57;68;75; 76;82;87	**E**	-	0	2	2	0	-	no statement possible (flight > 5km ?)
18346	*Laccobius* sp.	x			3		x			9;48;75;82;88	**E**	0	7	4	11	9	1.25	yes
5795	*Libellula depressa*	x			3		x			2;7;40;57;75;82;94	**E**	0	0	2	2	4	0.5	yes
19463	*Limnephilus affinis/incisus*	x			3		x	x	x	6;49;58;63;75;82;86	**E**	0	0	0	0	2	0	no statement possible
5817	*Limnephilus auricula*	x			3		x	x		49;58;75;78;81;82;86	**E**	0	2	0	2	2	1	no statement possible
5831	*Limnephilus griseus*	x			3		x	x		49;58;63;75;81;82	**E**	0	0	0	0	2	0	no statement possible

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
5837	*Limnephilus lunatus*	x			3		x	x		49;58;75;82;86	**E**	0	13	11	24	16	1.57	yes
5841	*Limnephilus rhombicus rhombicus*	x			3		x			49;58;75;63;82	**E**	0	0	0	0	9	0	no statement possible
5845	*Limnephilus stigma*	x			3		x	x		49;58;63;75;82	**E**	0	0	7	7	4	1.5	yes
6062	*Mystacides azurea*	x			2*		x	x		9;10;51,61;75;78;86	**E**	2.1	0	0	0	2	0	no statement possible
18467	*Nebrioporus elegans*	x			3	x				9;20;46;75;82	**E**	0	0	2	2	0	-	no statement possible (flight > 5km ?)
6403	*Pilaria* sp.	x			1					82;88	**E**	0	7	4	11	36	0.31	no --> change from D to E
18649	*Platambus maculatus*	x			3					9;18;20;39;46;52;75;82	**E**	2.2	2	0	2	2	1	no statement possible
6667	*Pyrrhosoma nymphula*	x			3		x			7;9;26;27;57;62;73;75;76;82; 84;93	**E**	2	0	4	4	4	1	yes
6795	*Rhypholophus* sp.	x			3					21;82	**E**	0	0	0	0	2	0	no statement possible
6911	*Stenophylax permistus*	x			3	x				75;82	**E**	-	0	0	0	2	0	no statement possible
8761	*Stratiomyiidae* gen. sp.	x			3		x			9;75;82	**E**	0	2	4	7	4	1.5	yes
6972	*Tanypodinae* gen. sp.	x			2					6;7;9;49	**E**	0	9	13	22	56	0.4	yes
6977	*Tanytarsini* gen. sp.	x			3					7;9;49;82	**E**	0	9	2	11	42	0.26	yes
7077	*Tipula* sp.	x			1*		x			9;82	**E**	0	0	4	4	11	0.4	no --> change from D to E
10370	Chaetopterygini/Stenophylacini Gen. sp.	x									**x**	0	7	0	7	27		-
8491	Corixidae gen. sp.	(x)									**x**	0	0	0	0	4		-
5299	*Gerris lacustris*	(x)									**x**	-	2	2	4	0		-
12529	*Helophorus* sp.	x									**x**	0	0	0	0	2		-
5545	*Hydrometra gracilenta*	(x)									**x**	-	0	2	2	0		-
5546	*Hydrometra stagnorum*	(x)									**x**	-	2	2	4	0		-

ID ART	Taxon	Winged	Habitat specialist	Pref. crenal	Dispersal capability	Rheophil	Limnophil	Warm stenotherm	Pot. pre. large rivers	References	**Dispersal class**	SI	FR_RC [%]	FR_RU [%]	FR_sum [%]	FR_S B,SO[%]	RQ	Validation: dispersal class confirmed in dispersal map
13126	Limnephilini gen. sp.	x									**x**	0	16	11	27	47		-
8483	Limoniidae gen. sp.	x									**x**	0	0	0	0	11		-
6118	*Nepa cinerea*	(x)									**x**	0	2	0	2	7		-
9321	Rhagionidae gen. sp.	x									**x**	0	0	0	0	2		-
8485	Tabanidae gen. sp.	x									**x**	0	0	0	0	11		-
7150	*Velia* sp.	(x)									**x**	-	2	0	2	4		-

References Table A2:

Amann E., Brandstetter C.M. & Kapp A. (1994) Käfer am Wasser. Gattungsschlüssel der (semi) aquatischen Käfer Mitteleuropas. Verlag des ersten Vorarlberger Coleopterologischen Vereins, Wien.

Angelibert S. & Giani N. (2003) Dispersal characteristics of three Odonata species in a patchy habitat. Ecography, 36, 13−20.

Askew R.R. (1988) The dragonflies of Europe. Harley Books, Colchester.

Bagge P. (1995) Emergence and upstream flight of lotic mayflies and caddisflies (Ephemeroptera and Trichoptera) in a lake outlet, central Finland. Entomologica Fennica, 6, 91−97.

Bauernfeind E. (1990) Der derzeitige Stand der Eintagsfliegen-Faunistik in Österreich (Insecta: Ephemeroptera). Verhandlungen der Zoologisch-Botanischen Gesellschaft Österreich, 127, 61−82.

Bellmann H. (1987) Libellen. J. Neumann Verlag, Melsungen.

Bellmann H. (1988) Leben in Bach und Teich. Pflanzen und Wirbellose der Kleingewässer. Steinbach Naturführer. Mosaik Verlag, München.

Bis B. & Usseglio-Polatera P. (2004) STAR Deliverable N2. Species Traits Analysis. Available at: www.eu-star.at/pdf/Deliverable_N2.pdf.

Blanke D. (1990) Zur Ephemeroptera - Plecoptera- und Trichoptera-Fauna des südniedersächsischen Mittelgebirgsbaches Bewer bei Markoldendorf (Kreis Northeim). Braunschweiger Naturkundlicher Schriften, 3, 669−697.

Botosaneanu L. & Malicky H. (1978) Trichoptera. In: Limnofauna Europaea. Eine Zusammenstellung aller die Binnengewässer bewohnenden mehrzelligen Tierarten mit Angaben über ihre Verbreitung und Ökologie (Ed. J. Illies), pp. 333−359. Vol. 2. Stuttgart, New York, Amsterdam.

Braasch D. & Jakob U. (1976) Die Verwendung von Ephemeropteren (Insecta) der DDR als Indikatoren für Gewässergüte. Entomologische Nachrichten, 20, 101−111.

Brandstetter C.M. & Kapp A. (1995) Die Schwimmkäfer von Voralberg und Liechtenstein. (Haliplidae, Noteridae, Dytiscidae, Gyrinidae).Verlag des ersten Vorarlberger Coleopterologischen Vereins, Wien.

Brauckmann U. (1987) Zoozönologische und saprobiologische Beiträge zu einer allgemeinen regionalen Bachtypologie. Archiv für Hydrobiologie, 26, 1−355.

Brehm J. & Meijering M.P.D. (1990) Fließgewässerkunde. Einführung in die Limnologie der Quellen, Bäche und Flüsse. Biologische Arbeitsbücher. Quelle & Meyer Verlag, Heidelberg, Wiesbaden.

Brockhaus T. (1991) Das Vorkommen von Fließwasser -Libellen im Bezirk Chemnitz aus Sicht des Naturschutzes. Libellula, 10, 35−44.

Burmeister E.G. (1984) Auswertung der Beifänge aquatischer Wirbelloser (Macroinvertebrata), aquatischer Wirbeltiere (Vertebrata) und terrestrischer

Wirbelloser (Macroinvertebrata). Berichte Anliegen Natur, 8, 205−212.

Burmeister E.G. (1986) Die Wasserkäferfauna des Naturschutzgebietes Federsee – Eine

Auswertung bisheriger Beobachtungen (Coleoptera, Hydradephaga und aquat. Palpicornia). Naturschutz Landschaftspflege Baden-Württemberg, 61, 261−275.

Burmeister E.-G. & Reiss F. (1983) Die faunistische Erfassung ausgewählter Wasserinsektengruppen in Bayern (Eintagsfliegen, Libellen, Steinfliegen, Köcherfliegen, Zuckmücken). Informationsbericht des Bayerischen Landesamts für Wasserwirtschaft, 8, 1−194.

Bussler H. (1992) Faunistische Dokumentation über die Schwimmkäfer von Augsburg und

Umgebung (Coleoptera: Noteridae, Dytiscidae). Bericht der Naturforschenden Gesellschaft Augsburg, 53, 6−28.

Caspers N. (1980) Die Emergenz eines kleinen Waldbaches bei Bonn. Decheniana-Beihefte, 23, 1−175.

Caspers N. & Stiers H. (1977) Beitrag zur Kenntnis der Plecoptera der Eifel (Insecta:

Plecoptera). Decheniana, 130, 136−150.

Chaput-Bardy A., Lemaire C., Picard D. & Secondi J. (2008) In-stream and overland dispersal across a river network influences gene flow in a freshwater insect, Calopteryx splendens. Molecular Ecology, 17, 3496−3505.

Chaput-Bardy A., Grégoire A., Baguette M., Pagano A. & Secondi J. (2010) Condition and Phenotype-Dependent Dispersal in a Damselfly, Calopteryx splendens. PLoS ONE, 5, 1−7.

Chinery M. (1987) Pareys Buch der Insekten. Ein Feldführer der europäischen Insekten. Paul Parey Verlag, Hamburg, Berlin.

Conrad K.F., Willson K.H., Harvey I.F., Thomas C.J. & Sherratt T.N. (1999) Dispersal characteristics of seven odonata species in an agricultural landscape. Ecography, 22, 524−531.

Conze K.J., Menke N. & Olthoff M. (2011) Libellen und Klimawandel in Nordrhein-Westfalen. Ergebnisse einer Studie zu Folgen des Klimawandels am Beispiel der Gestreiften Quelljungfer und der Arktischen Smaragdlibelle. Natur in NRW, 4, 20−26.

Dedecker A., van Melckebeke K., Goethals P. & de Pauw N. (2004) Development of migration models for macroinvertebrates in the zwalm river basin as a tool in river assessment and restoration management. Communications in Applied Biological Sciences, 69, 1−4.

Delette Y.R. & Morvan N. (2000) Dispersal of aquatic Chironomidae (Diptera) in agricultural landscapes. Freshwater Biology, 44, 399−411.

Deutscher Rat für Landespflege (2009) Verbesserung der biologischen Vielfalt in Fließgewässern und ihren Auen. Schriftenreihe des deutschen Rates für Landespflege, 92, 1−159.

Donath H. (1989) Verbreitung und Ökologie der Zweigestreiften Quelljungfer, Cordulegaster boltoni (Donovan, 1807), in der DDR (Insecta, Odonata: Cordulegasteridae). Faunistische Abhandlungen Staatliches Museum für Tierkunde, 16, 97−106.

Dreyer W. (1986) Die Libellen. Das umfassende Handbuch zur Biologie und Ökologie aller mitteleuropäischen Arten mit Bestimmungsschlüssel für Imagines und Larven. Gerstenberg Verlag, Hildesheim.

Ehlert T. (2009) Flugaktivität, Eiablage und Habitatbindung von Kocherfliegen (Trichoptera) an Fließgewässern. Essener Ökologische Schriften, 27, 1−169.

Elliott J.M. (2008) Ontogenetic shifts in drift periodicity and benthic dispersal in elmid beetles. Freshwater Biology, 53, 698−713.

Engelhardt W. (1986) Was lebt in Tümpel, Bach und Weiher? – Pflanzen und Tiere unserer Gewässer in Farbe; eine Einführung in das Leben der Binnengewässer. Franckh´sche Verlagshandlung (Kosmos Naturführer), Stuttgart.

Göthberg A. (1973) Dispersal of lotic Trichoptera from anorth Swedish stream. Aquilo - Series Zoologica, 14, 99−104.

Graf W., Grasser U. & Waringer J. (1995) Trichoptera (Köcherfliegen). In: Fauna Aquatica Austriaca. Katalog zur autökologischen Einstufung aquatischer Organismen Österreichs (Ed. O. Moog). pp.1−41. Herausgegeben im Auftrag des Bundesministeriums für Land- und Forstwirtschaft, Wien.

Havelka P. (1992) Rote Liste gefährdeter Gnitzen (Ceratopogonidae) Bayerns. Schriftenreihe des Bayerischen Landesamts für Umweltschutz, 111, 173−175.

Hebauer F. (1974) Über die ökologische Nomenklatur wasserbewohnender Käferarten (Coleoptera). Nachrichtenblatt der Bayerischen Entomologen, 23, 87−92.

Heidemann H. & Seidenbusch R. (1993) Die Libellenarten Deutschlands und Frankreichs. Handbuch für Exuviensammler. Verlag E. Bauer, Keltern.

Holm A. (1989) Ökologischer Bewertungsrahmen Fließgewässer (Bäche) für die Naturräume der Geest und des Östlichen Hügellandes in Schleswig-Holstein. Landesamt für

Naturschutz und Landschaftspflege des Landes Schleswig-Holstein, Kiel.

Horion A. (1960) Bemerkungen zur Faunistik der württembergischen Käfer. II. Haliplidae bis Scaphididae. Jahresheft des Vereins für vaterländische Naturkunde in Württemberg, 115, 315−329.

Illies J. & Botosaneanu L. (1963) Problèmes et méthodes de la classification et de la zonation ècologique des eaux courantes, considerées surtout du point de vue faunistique. Internationale Verhandlungen für theoretische und angewandte Limnologie, 12, 1−57.

Jacobs W. & Renner M. (1988) Biologie und Ökologie der Insekten. G. Fischer Verlag, Stuttgart.

Janecek B. & Contreras R. (1995) Chironomidae – Unterfamilie Chironominae. In: Fauna Aquatica Austriaca. Katalog zur autökologischen Einstufung aquatischer Organismen Österreichs (Ed. O. Moog). pp.1−13. Herausgegeben im Auftrag des Bundesministeriums für Land- und Forstwirtschaft, Wien.

Kehl S. & Dettner K. (2007) Flight Ability of the Adephagan Water Beetles (Coleoptera, Hydradephaga) of Germany. Entomologie heute, 19, 141−161.

Klausnitzer B. (1971) Beiträge zur Insektenfauna der DDR: Coleoptera – Helodidae. Beiträge zur Entomologie 21, 477−494.

Klausnitzer B. (1984) Käfer in und am Wasser. Ziemsen Verlag, Wittenberg.

Klausnitzer B., Jacob U., Jarisch O., Joost W., Klima F. & Peters G. (1978) Wasserinsekten. Kulturbund der DDR, Zentrale Kommission Natur und Heimat des Präsidialrates, Zentraler Fachausschuss Entomologie, Bautzen.

Klein R. (1984) Einfluß der Gewässergüte und der Wasservegetation auf Vorkommen und Abundanz von Calopteryx splendens, Platycnemis pennipes und Ischnura elegans, an sauerländischen Fließgewässern. Libellula, 3, 7−17.

Klima F. (1994) Die aktuelle Gefährdungssituation der Köcherfliegen Deutschlands (Insecta, Trichoptera). Natur und Landschaft, 69, 511−518.

Koch K. (1989) Die Käfer Mitteleuropas. Ökologie. Verlag Goecke & Evers, Krefeld.

Kuusela K. & Huusko A. (1996) Post-emergence migration of stoneflies (Plecoptera) into the nearby forest. Ecological Entomology, 21,171−177.

Lehmann U. (1971) Die Chironomiden der Fulda. Systematische, ökologische und faunistische Untersuchungen. Archiv für. Hydrobiolologie – Supplement, 37, 466−555.

Ludwig H.W. (1989) Tiere unserer Gewässer: Merkmale, Biologie, Lebensraum und Gefährdung. BLV-Verlag, München, Wien, Zürich.

Madsen B.L., Bengtson J. & Butz I. (1973) Observations on upstream migration by imagines of some Plecoptera and Ephemeroptera. Limnology and Oceanography, 18, 678−681.

Maibach A. & Meier C. (1987) Verbreitungsatlas der Libellen der Schweiz (Odonata, mit roter Liste). Documenta Faunistica Helveticae, 4, 1−228.

Malicky H. (1987) Anflugdistanz und Fallenfangbarkeit von Köcherfliegen (Trichoptera) bei Lichtfallen. Jahresbericht der Biologischen Station Lunz, 10, 140−157.

Malzacher P. (1981) Beitrag zur Insekten-Faunistik Südwestdeutschlands: Ephemeroptera – Eintagsfliegen. Mitteilungen Entomologischer Verein Stuttgart, 16, 41−72.

Masters Z., Petersen I., Hildrew A.G. & Ormerod S.J. (2007) Insect dispersal does not limit the biological recovery of streams from acidification. Aquatic Conservation: Marine and Freshwater Ecosystems, 17, 375−383.

Mey W. (1993) Kommentiertes Verzeichnis der Köcherfliegen (Trichoptera) der Länder Berlin und Brandenburg. In: Fauna in Berlin und Brandenburg. Schmetterlinge und Köcherfliegen (Eds M. Gerstberger & W. Mey), pp. 135−145. Berlin.

Meyer D. (1987) Makroskopisch-biologische Feldmethoden zur Wassergütebeurteilung von Fließgewässern. AG Limnologie u. Gewässerschutz e.V. & Bund für Umwelt und Naturschutz, Hannover.

Moog O. (1995) Fauna Aquatica Austriaca. Katalog zur autökologischen Einstufung aquatischer Organismen Österreichs. Wasserwirtschaftskataster, Bundesministerium für Land- und Forstwirtschaft, Wien.

Müller-Liebenau I. (1969) Revision der europäischen Arten der Gattung Baetis Leach, 1985 (Insekta Ephemeroptera). Limnologische Station Niederrhein in der Max-Plank-Gesellschaft, Krefeld-Hülserberg.

Petersen I, Winterbottom J.H., Orton S, Friberg N., Hildrew A.G., Spiers D.C. & Gurney W.S.C. (1999) Emergence and lateral dispersal of adult Plecoptera and Trichoptera from Broadstone Stream, UK. Freshwater Biology, 42, 401−416.

Pitsch T. & Weinzierl A. (1992) Rote Liste gefährdeter Köcherfliegen (Trichoptera) Bayerns. Schriftenreihe des Bayerischen Landesamts für Umweltschutz, 111, 201−205.

Poff N.L., Olden J.D., Vieira N.K.M., Finn D.S., Simmons M.P. & Kondratieff B.C. (2006) Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. Journal of the North American Benthological Society, 25, 730−755.

Reiff N. (1993) Aquatische Makroinvertebraten der ICE-Trassenkartierung; Kommentierte Artenliste. In: Ökologische Typisierung der aquatischen Makrofauna (Eds U. Schmedtje & M. Colling). p. 543. Bayerisches Landesamt für Wasserwirtschaft, München.

Reiter C. (1993) Beobachtungen zur Autökologie von Ischnura pumilio, Charpentier 1825, und Ischnura elegans, v.d. Linden 1820, in der Umgebung von Freising (Insecta, Odonata). Dragonfly Reseach, 4, 1−151.

Samietz R. (1989) Die Chironomiden – Emergenz von Spitter und Vesser (Diptera). Gesamtemergenz; Diamesinae – Gothaer Emergenz- Untersuchungen im Biosphärenreservat Vessertal Nr. 13. Verhandlungen IX. SIECC Gotha, 1986: 321−324.

Sauer F. (1987) Fliegen und Mücken. Fauna Verlag, Karlsfeld.

Sauer F. (1988) Wasserinsekten. Fauna Verlag, Karlsfeld.

Schmedtje U. (1995) Ökologische Grundlagen für die Beurteilung von Ausleitungsstrecken – Beziehungen zwischen der sohlnahen Strömung, dem Gewässerbett und dem Makrozoobenthos. Schriftenreihe des Bayerischen Landesamts für Wasserwirtschaft, 25, 1−156.

Schmedtje U. & Kohmann F. (1992) Bestimmungsschlüssel für die Saprobier-DIN-Arten (Makroorganismen). Informationsberichte des Bayerischen Landesamts für Wasserwirtschaft, 88, 1−274.

Schmedtje U. & Zwick H. (1992) Diptera. Informationsberichte des Bayerischen Landesamts für Wasserwirtschaft, 88, 229−260.

Schmedtje U. & Colling M. (1996) Ökologische Typisierung der aquatischen Makrofauna. Bayerisches Landesamt für Wasserwirtschaft, München.

Schmidt-Kloiber A. & Hering D. (2012) www.freshwaterecology.info - the taxa and autecology database for freshwater organisms. Available at: www.freshwaterecology.info (Version 5.0).

Schorr M. (1990) Grundlagen zu einem Artenhilfsprogramm Libellen der Bundesrepublik Deutschland. Societas Internationalis Odonatologica- Ursus Scientific Publishers, Bilthoven (NL).

Schutte G., Reich M. & Plachter H. (1997) Mobility of the rheobiont damselfly Calopteryx splendens (Harris) in fragmented habitats (Zygoptera: Calopterygidae). Odonatologica, 26, 317−327.

Seitz G. & Weinzierl A. (1991) Köcherfliegen. In: Ökologische Zustandserfassung und Beweissicherung Untere Isar zwischen Ettingen und Isarmünden (Zus.-fas.) (Ed. J. Pfadenhauer), pp. 1−101. TU München, Institut für Landespflege, Freising-Weihnstephan.

Sode A. & Wiberg-Larsen P. (1993) Dispersal of adult Trichoptera at a Danish forest brook. Freshwater Biology, 30, 439−446.

Stettmer C. (1996) Colonization and dispersal patterns of banded (Calopteryx splendens) and beautiful demoiselles (C. virgo) (Odonata: Calopterygidae) in south-east German streams. European Journal of Entomology, 93, 579−593.

Svensson B.W. (1974) Population movements of adult Trichoptera at a South Swedish stream. OIKOS, 25, 157−175.

Thiele V., Mehl D., Berlin A. & Huijsson L. (1998) Untersuchungen zum Gegenstromwanderungsverhalten aquatischer und zum Gegenstromflug merolimnischer Evertebraten im Bereich von Fischaufstieganlagen in Mecklenburg-Vorpommern (Deutschland). Limnologica, 28, 167−182.

Thomes A. (1987) Auswirkungen anthropogener Veränderungen eines norddeutschen Tieflandbaches auf die Libellenfauna. Limnologica, 18, 253−268.

Timm T. (1993) Einzigartige Biozönose – Erhalt des gering belasteten Wienbaches Herausforderung für den Naturschutz. LÖLF-Mitteilungen, 93, 19−23.

Tobias W. & Tobias D. (1981) Trichoptera Germanica. Bestimmungstafeln für die deutschen Köcherfliegen. Teil I- Imagines. Courier Forschungsinstitut Senckenberg, 49, 1−672.

Van Noordwijk M. (1978) A mark-recapture study of coexisting zygopteran populations. Odonatologica, 7, 353−374.

Vieira N.K.M, Poff N.L., Carlisle D.M., Moulton II S.R., Koski M.L. & Kondratieff B.C. (2006) A database of lotic invertebrate traits for North America: U.S. Geological Survey Data Series 187. Available at: http://pubs.water.usgs.gov/ds187-Vieira_2006_InvertTraits Table_v1.txt.

Wagner R. (1978) Ptychopteridae. In: Limnofauna Europaea. A checklist of the animals inhabiting European inland waters with accounts of their distribution and ecology (except Protozoa), (Ed. J. Illies), p. 386. Verlag Gustav Fischer, Stuttgart, New York.

Wagner R. (1992) Rote Liste gefährdeter Tanzfliegen (Empididae, Unterfamilien Hemerodromiinae und Clinocerinae) Bayerns. Schriftenreihe des Bayerischen Landesamts für Umweltschutz, 111, 176−177.

Wagner R. & Gerecke R. (2008) Tanzfliegen (Diptera: Empididae) aus Quellen im Nationalpark Gesäuse (Österreich). Lauterbornia, 63, 77−82.

Ward L. & Mill P. (2007) Long range movements by individuals as a vehicle for range expansion in Calopteryx splendens (Odonata: Zygoptera). European Journal of Entomology, 104, 195–198.

Waringer J.A. (1986) Beitrag zur Kenntnis der Libellenfauna von Wien und Niederösterreich. Libellula, 5, 47−64.

Waringer J.A. (1989) Gewässertypisierung anhand der Libellenfauna am Beispiel der Altenwörther Donauau (Niederösterreich). Natur und Landschaft, 64, 389−392.

Weinzierl A. & Dorn A. (1989) Stein- und Köcherfliegen – Nachweise entlang der Münchener Isar (Insecta: Plecoptera, Trichoptera). Lauterbornia, 1, 6−22.

Zahner R. (1960) Über die Bindung der Mitteleuropäischen Calopteryx-Arten (Odonata, Zygoptera) an den Lebensraum des Strömenden Wassers II. Der Anteil der Imagines an der Biotopbindung. Internationale Revue der gesamten Hydrobiologie und Hydrographie, 45, 101−123.

Aus datenschutzrechtlichen Gründen ist der Lebenslauf in der Online-Version nicht enthalten.

Publikationen

Winking C., Lorenz, A.W., Sures B. & Hering D. (in prep.) Restoration of a river system in an urban area: towards the Good Ecological Potential of former sewage channels.

Winking C., Lorenz, A.W., Sures B. & Hering D. (under review) Succession of benthic invertebrate assemblages in restored former sewage channels. Aquatic Sciences.

Winking C., Lorenz, A.W. & Hering D. (2015) Die naturnahe Umgestaltung des Emschersystems und das Ökologische Potenzial: zur Entwicklung aquatischer Biodiversität in ehemaligen Abwasserrinnen. Natur und Landschaft, (accepted).

Winking C., Lorenz, A.W., Sures B. & Hering D. (2014) Recolonisation patterns of benthic invertebrates: a field investigation of restored former sewage channels. Freshwater Biology, 59, 1932–1944.

Winking C., Korte, T. & Lorenz, A.W. (2013): Die Wiederbesiedlung urbaner Fließgewässer in einem Teileinzugsgebiet der Emscher nach erfolgten ökologischen Verbesserungen. Korrespondenz Wasserwirtschaft, 6, 310–317.

Essen, 10. April 2015

Eidesstattliche Erklärungen

Erklärung:

Hiermit erkläre ich, gem. § 6 Abs. (2) f) der Promotionsordnung der Fakultäten für Biologie, Chemie und Mathematik zur Erlangung der Dr. rer. nat., dass ich das Arbeitsgebiet, dem das Thema „***Ecological evaluation of restored former sewage channels in the urbanised Emscher catchment***" zuzuordnen ist, in Forschung und Lehre vertrete und den Antrag von Caroline Winking befürworte.

Essen, den ___________________ ____

Unterschrift eines Mitglieds der Universität Duisburg-Essen

Erklärung:

Hiermit erkläre ich, gem. § 7 Abs. (2) c) + e) der Promotionsordnung der Fakultäten für Biologie, Chemie und Mathematik zur Erlangung des Dr. rer. nat., dass ich die vorliegende Dissertation selbständig verfasst und mich keiner anderen als der angegebenen Hilfsmittel bedient habe.

Essen, den ___________________ _________________________________

Unterschrift der Doktorandin

Erklärung:

Hiermit erkläre ich, gem. § 7 Abs. (2) d) + f) der Promotionsordnung der Fakultäten für Biologie, Chemie und Mathematik zur Erlangung des Dr. rer. nat., dass ich keine anderen Promotionen bzw. Promotionsversuche in der Vergangenheit durchgeführt habe und dass diese Arbeit von keiner anderen Fakultät abgelehnt worden ist.

Essen, den ___________________ ____________________________

Unterschrift der Doktorandin